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An assessment of tree, snag, and downed wood residuals in boreal fires

in relation to Ontario's policy directions for emulating natural forest disturbance

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Abstract

This study focused on assessing abundance, spatio-temporal variability, and causal factors of post-fire residuals as relevant to directions in Ontario's *Forest Management Guide for Natural Disturbance Pattern Emulation* (NDPE guide). We assessed residual trees, snags, and downed wood in 11 fires from 3 ecoregions in boreal Ontario using high-resolution airborne photographs (660 plots) as well as field samples (50 plots). Photo-plots were used to assess the abundance of residuals and field plots were used to quantify photo interpretation error as well as to monitor residual dynamics over three years. Abundance of residuals immediately post-fire, in all categories, varied widely both within and among fires. Occurrence of residual trees was spatially sporadic while snags were omni present. Expected residual tree abundance was considerably lower than that for snags. However, abundance of residual trees changed rapidly, with most changes occurring between the first and second year after fire. Due to residual tree mortality, the complement of snag and downed wood residuals increased in time. After three years, abundance of large diameter residual trees was very low, and congruent with directions provided in the NDPE guide for retaining residual trees post-harvest. Local fire intensity appeared to be the most important global determinant of occurrence of residual trees, but with an inverse relationship. Our results do not support the hypotheses that pre-burn forest cover and site conditions are reliable global predictors of residual tree occurrence.

Résumé

L'étude a été réalisée pour examiner le matériel forestier (abondance, variabilité spatiotemporelle et facteurs causatifs) laissé après un incendie de forêt par rapport aux directives provinciales exposées dans le guide intitulé *Forest Management Guide for Natural Disturbance Pattern Emulation* (guide NDPE). Nous avons examiné les arbres sur pied, les arbres abattus et les chicots laissés après 11 incendies dans trois écorégions de l'Ontario boréal. Nous avons utilisé à cette fin des photographies aériennes haute définition (660 parcelles) et des examens sur place (50 parcelles). Les photographies et les parcelles expérimentales ont servi à déterminer le degré d'abondance du matériel forestier résiduaire. Les parcelles expérimentales ont servi à quantifier les erreurs liées à l'interprétation des photographies et à surveiller l'évolution du matériel résiduaire sur une période de trois ans. Le degré d'abondance du matériel résiduaire tout de suite après un incendie, pour toutes les catégories, varie énormément au sein des incendies et entre ceux-ci. La présence d'arbres résiduaires était irrégulière dans l'espace, tandis que les chicots étaient omniprésents. Le nombre prévu des arbres résiduaires était considérablement plus faible que celui des chicots. Toutefois, le nombre des arbres résiduaires a changé rapidement, la plupart des changements ayant eu lieu entre la première année et la deuxième année après l'incendie. En raison du taux de mortalité des arbres résiduaires, le pourcentage représenté par les chicots et les arbres abattus a augmenté progressivement. Après trois ans, le nombre d'arbres résiduaires d'un grand diamètre était très faible. Il concordait avec les directives exposées dans le guide NDPE en ce qui concerne le nombre d'arbres laissés sur pied après une récolte. L'intensité des incendies semble être le plus important facteur déterminant de la présence d'arbres résiduaires, mais le rapport est inversé. Nos résultats ne soutiennent pas les hypothèses selon lesquelles le couvert forestier et les caractéristiques d'un site avant un incendie seraient de bons indices globaux de la présence d'arbres résiduaires.

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Introduction

Background

Fires leave behind remnants of the original forest as live stems and standing dead and downed stems. This remnant structure is collectively referred to as post-fire *residuals*, and these are thought to be important in the ecological functions of post-fire ecosystems. Thus, retaining residuals during forest harvest to create remnant structure similar to that created by natural fire disturbances may help to sustain forest ecological processes and thus conserve biodiversity (Hunter 1990).

Given the current emphasis on emulating natural forest disturbances as a formal forest management goal, knowledge of post-fire residual structure has become an important aspect of planning forest harvests. This is especially evident in Canada, where various formal approaches have been adopted for retaining unharvested forest structure as post-harvest residuals to meet goals for the emulation of natural disturbance (e.g., BCMF 1995, ASRD 2006, Jetté 2007).

In Ontario, emulating natural forest disturbances and landscape patterns is a principle in the *Crown Forest Sustainability Act* (Statutes of Ontario 1995), which has led to the development of forest management policies that guide forest harvesting practices based on patterns of natural disturbance. The *Forest Management Guide for Natural Disturbance Pattern Emulation* (the NDPE guide, OMNR 2001), which has been applied in Ontario since 2003, specifies directions and provides standards and guidance that help forest managers to emulate fire disturbances during forest harvest, including the amount of residual structure to be retained. The guide directs forest managers to leave specific amounts of residual structure in the form of individual trees and snags, as well as an unspecified amount of downed woody material, during timber harvesting.

In 2003, the Ontario Ministry of Natural Resources (MNR) was requested to assess the effectiveness of the directions provided in the NDPE guide (Condition 39c of the Declaration Order MNR-71 [OEAB 2003] under the *Environmental Assessment Act*). Under this mandate, a series of multi-scale scientific studies (described by Perera and Buse [2006]) was initiated in 2005 to improve the understanding of the characteristics of natural fire regimes in Ontario's fire-driven landscapes. One aspect of this series of studies was to examine and reduce the uncertainties associated with the NDPE guide directions for leaving residual trees, residual snags, and downed wood.

As a first step in reducing these uncertainties, we documented the state of published knowledge about post-fire residuals in boreal forests of North America, specifically as relevant to Ontario's NDPE guide directions (Perera et al. 2007). We focused on determining what information is available in the published literature about the abundance and variability of post-fire residual trees, residual snags, and downed woody debris to assess the gaps in knowledge. From this review, it was evident that although the published knowledge on post-fire residuals provides estimates of their abundance and extent, the standards used to define, assess, and quantify these residuals are inconsistent and sometimes ambiguous, making it difficult to generalize the information. As well, knowledge of the variability in residuals and its sources is sparse, even though this information is needed to develop specific management objectives and practices based on the broad policies.

Goal of this report

By understanding the characteristics of post-fire residual structure under natural conditions, forest policymakers can provide better strategic guidance to help forest managers emulate natural patterns of fire disturbance during forest harvesting, and to make better tactical decisions about how to retain post-harvest residual structure that emulates fire disturbances. Thus, the objectives of the study reported here were to characterize the abundance and variability in residual stand structure within unsuppressed boreal forest fire events to understand the extent of post-fire residual structure (residual trees, residual snags, and downed wood), its within- and among-fire variability, as well as how it changes during the first few years following the fire.

Specifically, we examined unsuppressed fires in boreal Ontario to:

- Describe the extent and variability of the residual structure that results from forest fires
- Document the immediate post-fire changes that occur in residual structure

We relate these results to what has been reported in the literature and the directions provided in Ontario's NDPE guide.

This report presents results of a study of post-fire residual structure in boreal forests with the goal of understanding its extent and variability as well as short-term changes.

Methods

Study area

The study area was the boreal forest region and Great Lakes–St. Lawrence transitional zone of Ontario (Rowe 1972), which is dominated by black spruce (*Picea mariana* [Mill.] B.S.P.), jack pine (*Pinus banksiana* Lamb.), trembling aspen (*Populus tremuloides* Michx.), birch (*Betula* spp.,), and balsam fir (*Abies balsamea* [L.] Mill.), with significant red pine (*Pinus resinosa* Ait.) and white pine (*Pinus strobus* L.) components in the more southern parts.

The overall forest disturbance regime is fire-driven but insect epidemics and blowdown are also common. The land base is primarily Crown land managed extensively for timber and other values and much of it has been harvested at least once during the past century. Since 2003, forest harvesting has been guided by directions in the NDPE guide (OMNR 2001), which specifies the amount of residual structure to be retained, with the intent of emulating fire disturbance.

Study fires

Fires to sample for post-fire residual structure were selected from the set of all fires that burned in boreal Ontario in 2005. That year, fire burned about 1% of boreal Ontario (37,985 ha), with a total of 1,045 fires reported. This is more than the average annual number of fires (868), but much less area was affected than the average of 98,942 ha per year reported for 1975-2006 (based on OMNR fire records). This indicates that there were many more small fires in 2005 than in the average year.

Fires were selected using the following criteria: availability of spatial data, including forest resource inventory data (pre-fire forest cover type, age, and density) and the stand's disturbance history; areas that were not disturbed within the last 30 years before the fire; and availability of fire perimeter and suppression activity information to indicate whether the entire fire or a large area within the fire had not been subjected to fire suppression. As well, a range of fire sizes was preferred. It was not our intent to sample representatively across all ecoregions. Eleven study fires were selected across the province and they happened to occur in three ecoregions (Figure 1).

The pre-fire forest cover (from MNR forest resource inventory [FRI]; OMNR 2000) varied among regions. Study fires in ecoregion 3W were mostly in black spruce forests with some jack pine, poplar (*Populus* spp.), white birch (*Betula papyrifera* Marsh.), and balsam fir. Of the study fires in ecoregion 3E, three were mostly spruce with poplar or pine, one was mostly jack pine with some spruce and white birch, and one was a mixedwood with spruce, pine, fir, poplar, larch (*Larix* spp.). Study fires in ecoregion 4W were in conifer mixedwoods with red pine, white pine, and jack pine and mixed hardwoods (data from the Quetico Forest Inventory 2000) (details provided in Figures 3-5). Pre-fire forest cover age distributions varied, with fires containing trees ranging in age from 60 to >140 based on the most recent FRI (details provided in Figures 3-5 below) (OMNR 2000).

Figure 1. *Locations of the 2005 fires (•) selected for sampling post-fire residual structure in three ecoregions (3W, 3E, 4W) in boreal Ontario.(Fire labels follow MNR convention.)*

The study fires ignited between 26 May and 6 August 2005 under a range of fire weather index values. Final fire sizes varied from 60 to 13,623 ha. Since our intent was to study natural fire dynamics, in most fires a subset of the fire area was sampled to avoid areas that had been subjected to suppression activities. These areas were located based on information from MNR's provincial fire management database and district fire reports. The areas in which suppression occurred ranged from none in fires in ecoregion 4W to most of the fire area, as was the case in fire TIM-019. Table 1 provides details about the study fires. Figure 2 shows an aerial view of the largest study fire (NIP-020). Detailed fire information and summaries of the fire reports for each fire are provided in Appendix 1.

***Table 1**. Locations, season, fire weather intensity, fire growth days, total fire size, and sampled fire area for the 11 study fires across boreal Ontario (detailed fire information and descriptions are provided in Appendix 1).*

Eco region	MNR fire code for the study fire	Latitude (° N)	Longitude (° W)	Fire season[1]	Fire weather intensity rating[1,2]	Fire growth days[1]	Total fire size[1] (ha)	Sampled fire area[3] (ha)
3W	NIP-020	48.273	-80.778	Summer	Extreme	4	13,623	6,734
	THU-030	48.293	-91.659	Summer	High	3	429	350
	THU-031	48.177	-91.371	Summer	High	4	1,428	1,229
	THU-067	50.545	-87.981	Summer	High	4	2,326	969
3E	COC-010	49.452	-81.131	Spring	Extreme	2	350	347
	TIM-005	50.581	-88.744	Spring	High	1	440	306
	TIM-007	50.293	-88.776	Spring	High	1	449	214
	TIM-019	50.176	-89.710	Summer	Extreme	5	3,200	146
	CHA-018	48.302	-82.922	Summer	Extreme	1	60	41
4W	FOR-014	47.767	-81.794	Summer	High	–	430	430
	FOR-024	48.190	-82.230	Summer	Moderate	–	213	213

[1] From MNR district fire reports

[2] Based on the OMNR fire weather index rating for fire intensity (low, <3.0; moderate, 3.0 to 9.9; high, 10.0 to 22.9; extreme, ≥23.0), which combines Initial Spread Index (expected rate of fire spread) and Buildup Index (fuel available for consumption) based on the system of Forestry Canada's Fire Danger Group (1992).

[3] The sample area is smaller that total fire size because areas where fire suppression or post-fire salvage logging occurred were excluded from the sample.

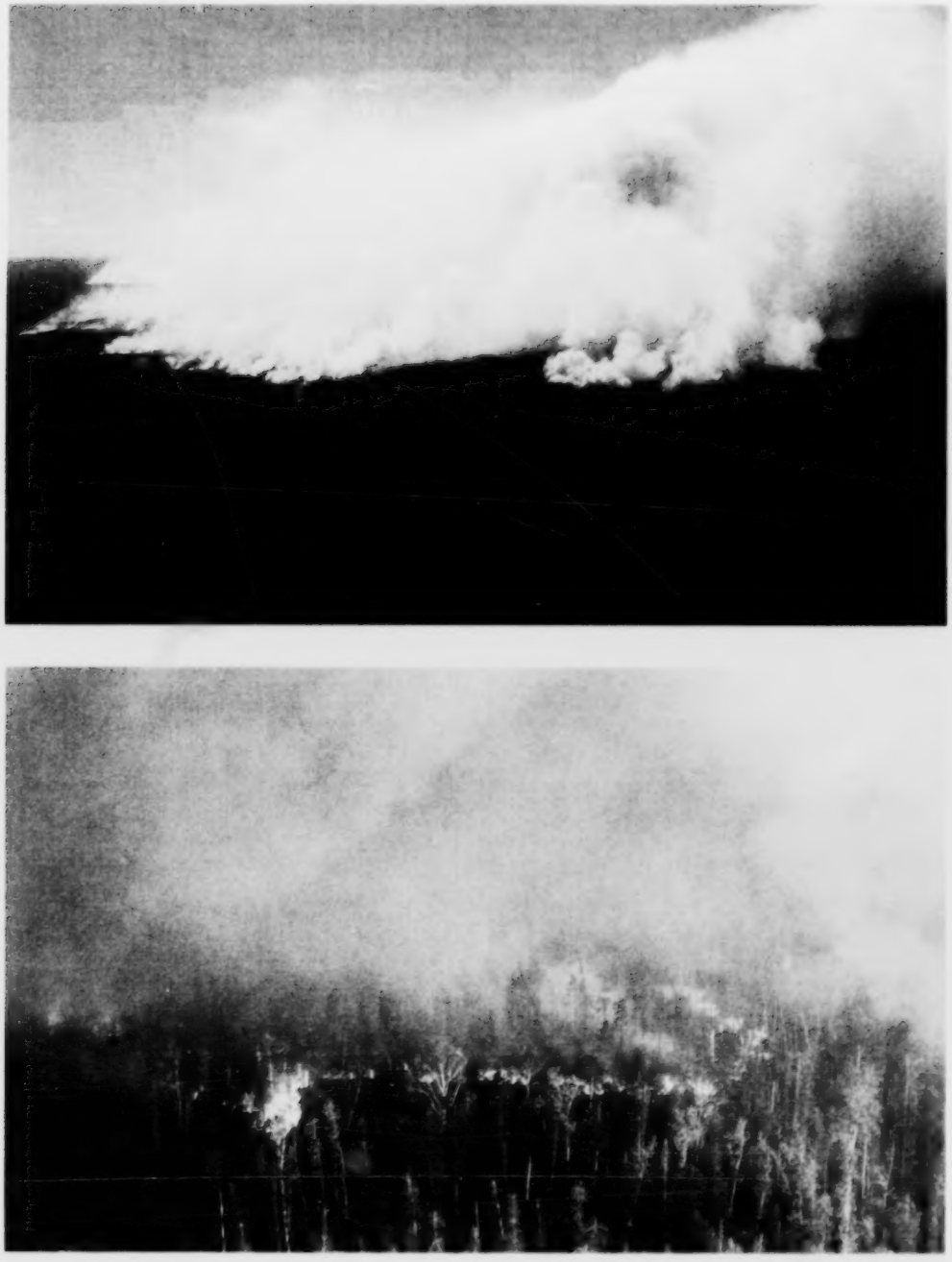


Figure 2. *Photographs of the NIP-020 fire burning on 20 June 2005. This was the largest fire we sampled.*

High-resolution photography sampling

Sample points

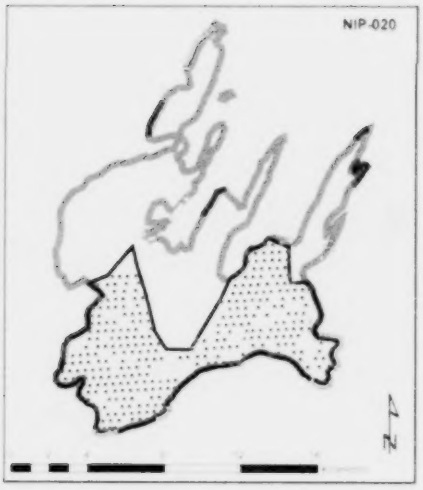
Post-fire residual structure in all study fires was sampled using high-resolution aerial photographs (HRPs). These photographs were taken in September 2005 (within 2 to 5 months after the fire) prior to deciduous leaf fall to allow us to distinguish between live and dead trees. Images of the post-fire stand structure were captured using a camera mounted on a helicopter from an altitude of approximately 100 m (for details of the HRP technology and sampling, see Routledge 2007). This method has been applied elsewhere, including for forest inventory; surveys of stand density, tree height, and crown area; and species identification within established or regenerating stands (e.g., Hall and Aldred 1992, Pitt 1994). It has also been used to estimate post-fire (40 to 60 years later) densities of residual trees and snags (DeLong and Tanner 1996).

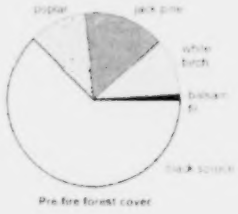
The HRP images were captured as systematic point samples (stereopairs) in the burned areas, oriented along a series of east–west transects across the area selected for sampling (and placed to avoid areas in which fire suppression and recent human disturbance had occurred, as well as unburned islands larger than 0.25 ha and water). The sample transects were spaced at intervals of 200 to 400 m within the fires, and sample points along transects were spaced a minimum of 300 m apart. Spacing increased with increasing fire size. Sample points were also positioned to ensure that they represented a range of distances from fire edges (the target minimum distance from the edge was 50 m).

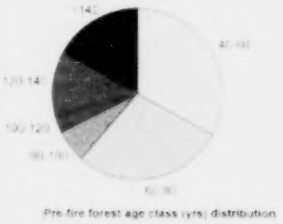
Sampling intensity varied among the fires depending on their size. Figures 3-5 show the locations of the HRP sample points within the study fires. In all, 739 points were sampled across the 11 study fires; of these, 660 images were interpreted (Table 2). The 79 images that were not interpreted fell in unburned forest, non-forested areas (e.g., wetland), or areas with anthropogenic disturbance (e.g., salvage logging or fire suppression activity). Figure 6 provides an example high-resolution photograph illustrating the level of detail of post-fire residual structure visible on the images.

***Table 2**. Sampling intensity using high-resolution photograph (HRP) across the 11 study fires in boreal and Great Lakes–St. Lawrence transitional zone in Ontario.*

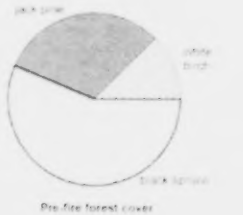
Study fire	HRP sample points (no.)	Sampling intensity (no. sample points/ha of study fire area)
NIP-020	309	0.05
THU-030	32	0.09
THU-031	84	0.07
THU-067	83	0.09
COC-010	31	0.09
TIM-005	27	0.09
TIM-007	20	0.09
TIM-019	17	0.12
CHA-018	7	0.17
FOR-014	32	0.07
FOR-024	18	0.08
Overall	**660**	**0.06**


NIP-020
N


white birch
Pre-fire forest cover


140+
120-140
100-120
80-100
Pre-fire forest age class (yrs) distribution

THU-030
N


white birch
Pre-fire forest cover

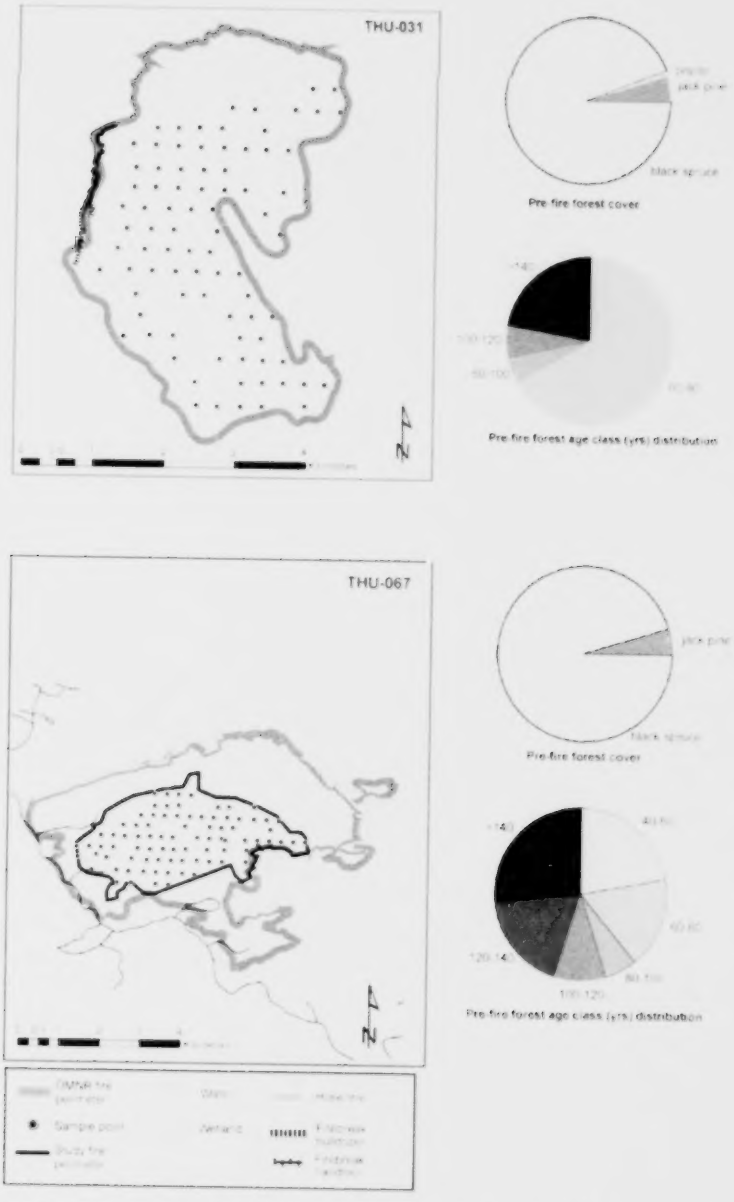
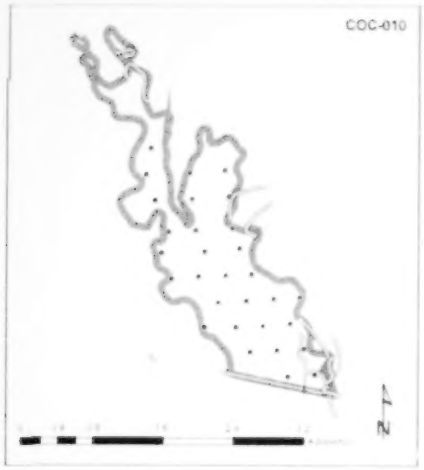
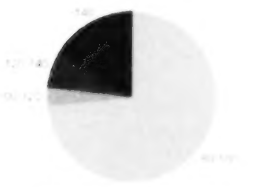


Figure 3. *Maps of study fires in the Northwest Region showing the pre-fire forest cover and age distribution, fire perimeters, fire suppression lines, the research study area, and the locations of the high-resolution photograph sample points (dots).*



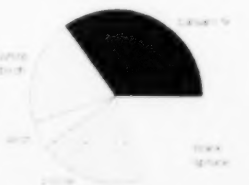
Pre-fire forest cover



Pre-fire forest age class (yrs) distribution

Pre-fire forest cover

Pre-fire forest age class (yrs) distribution



Pre-fire forest cover

Pre-fire forest age class (yrs) distribution

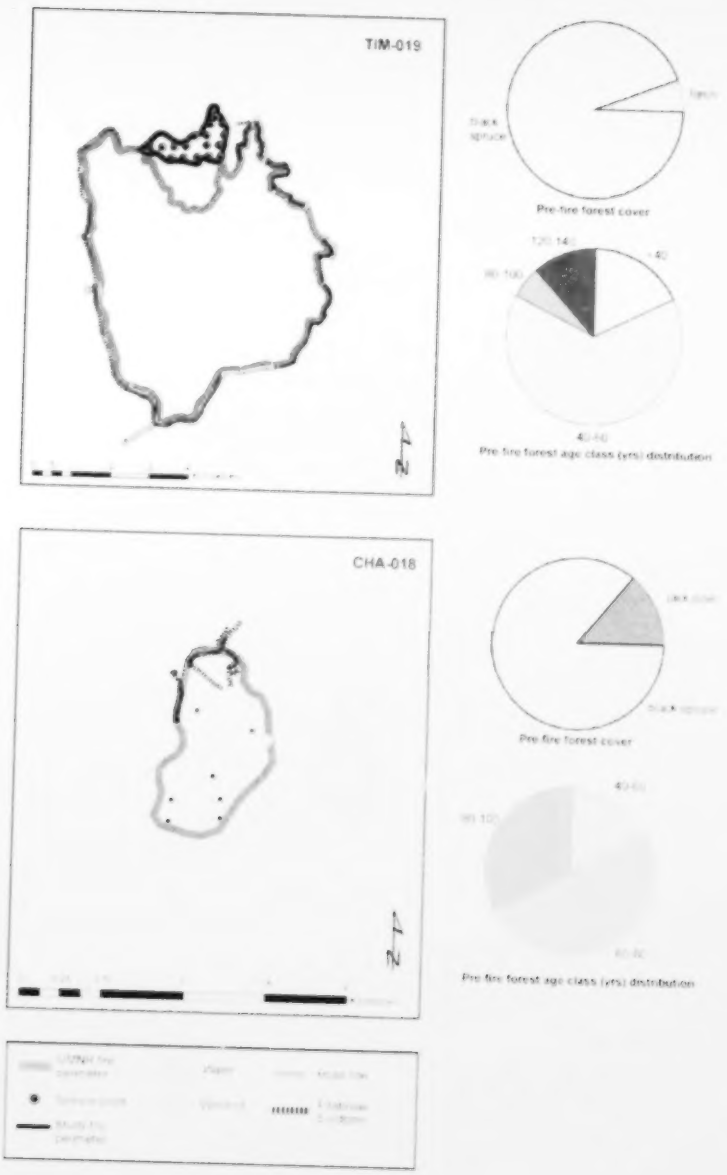


Figure 4. *Maps of study fires in the Northeast Region showing the pre-fire forest cover and age distribution, fire perimeters, fire suppression lines, the research study area, and the locations of the high-resolution photograph sample points (dots).*

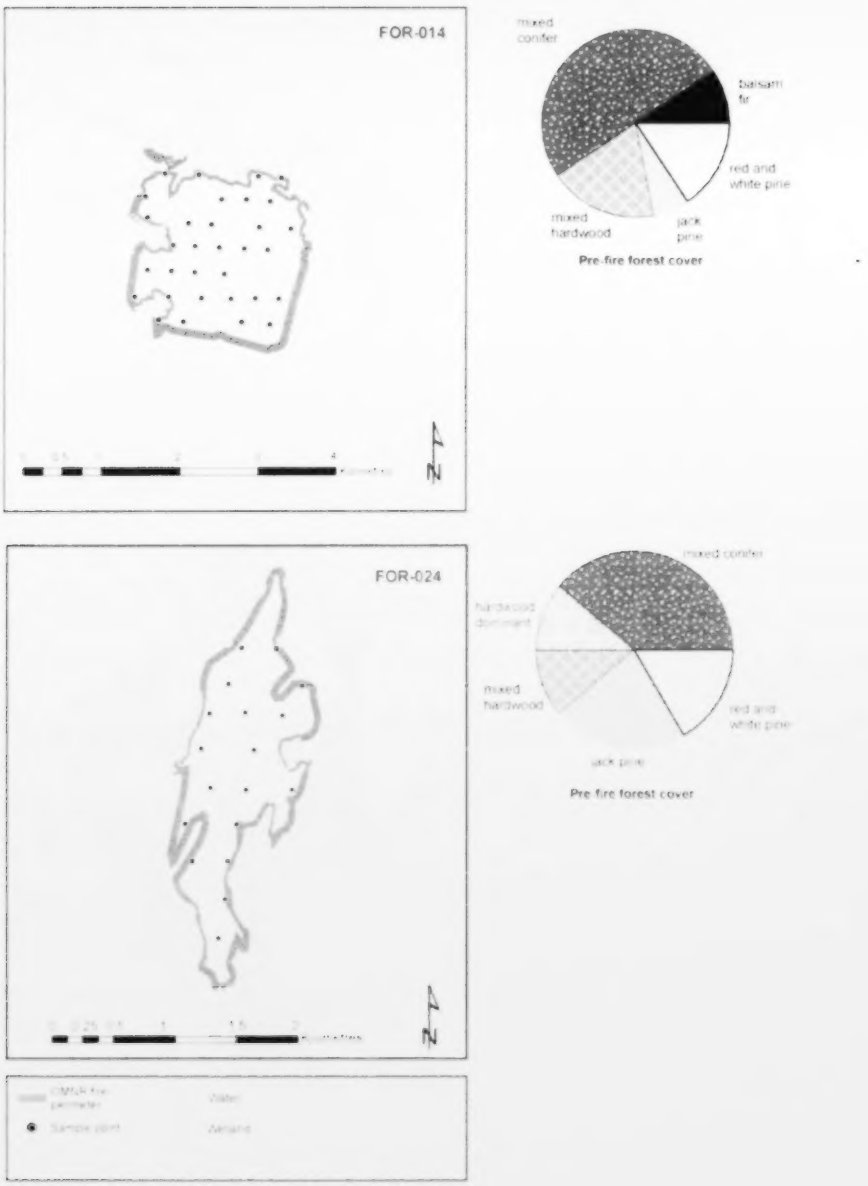


Figure 5. *Maps of study fires in the Great Lakes–St. Lawrence transitional zone showing the pre-fire forest cover, fire perimeters, fire suppression lines, the research study area, and the locations of the high-resolution photograph sample points (dots). (Forest age information was not available for these fires.)*

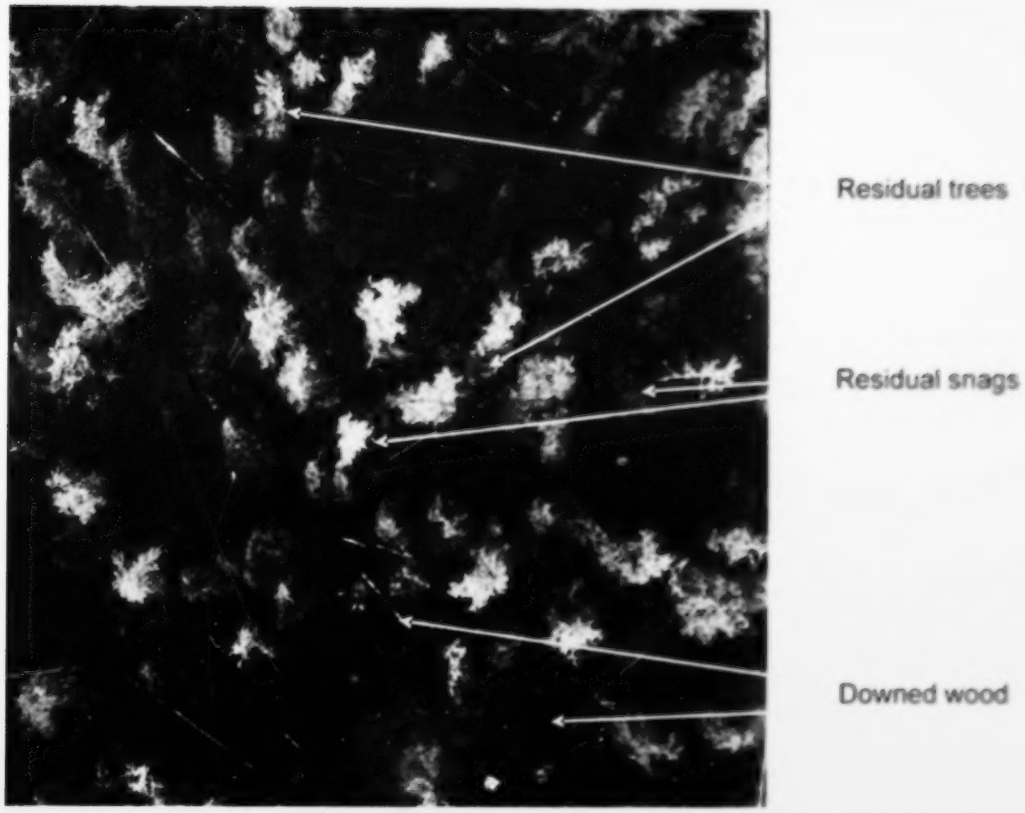


Figure 6. *Sample high-resolution photograph showing the level of detail visible for the residual structure. Residual trees have some green foliage remaining, residual snags are grey and brown, downed wood is lying on the surface.*

Photo interpretation

The stereo-pairs of the images were enlarged 2.45 times to produce 15.2 cm x 15.2 cm photos for interpretation. A circular plot, equivalent to 12.6 m in radius on the ground, centred on each enlarged HRP image, was selected for interpreting the stand-scale residual structure. This area (~500 m^2) was visually interpreted by a single interpreter to quantify the post-fire residual structure. Interpretation was semi-automated by using the RMAP software (adapted from Pitt 1994). We grouped residuals into three categories: residual trees, residual snags, and downed wood. Standing stems in each plot were interpreted first, starting at the top right and continuing around the plot, followed by the downed wood (Figure 7). Residual trees were defined as stems at least 2 m in height with green foliage present; residual snags were defined as stems with no green foliage, at least 2 m in height, and with a lean less than or equal to 45°; and downed wood was defined as stems that were either prone or had a lean greater than 45°, and that had at least 2 m of their length inside the interpreted area. Downed wood was further classified based on its diameter (<10.0 cm, 10.0-24.5 cm, >24.5 cm). For details of the photo interpretation, see Routledge (2007). The variables that were interpreted from photos for each category of residual are summarized in Table 3.

Post-fire residual structure categories used in this study:

Residual trees: *stems at least 2 m in height with green foliage present*

Residual snags: *stems at least 2 m in height, with no green foliage, and a lean less than or equal to 45°*

Downed wood: *stems either prone or with a lean greater than 45°, with at least 2 m of their length inside the plot*

Figure 7. *An example of how the interpreter marked the residual structure in a sample plot on a high-resolution photograph. Trees (green) and snags (blue) were marked first (on the crown), then downed wood was marked by diameter class (red, green, blue along the length of the stem). The plot boundary was determined by the RMAP software; the blue circle served only as a rough guide for interpreter.*

In interpreting the HRP images, two major sources of confounding occurred in these residual categories: (a) trees that died prior to the fire could not be separated from those that died during the fire and both were categorized as residual snags, and (b) snags and trees that had fallen before the fire could not be separated from those that fell during the fire and both were categorized as downed wood.

Table 3. *Summary of the post-fire structure categories and variables measured during interpretation of the sample plots in high-resolution photographs.*

Residual category and description	Variable interpreted	Interpreted details
Residual tree ≥2 m tall[1] green foliage present	Cover class	Tree species
	Canopy position	Dominant, codominant, intermediate, suppressed
	Height (m)	For each canopy position
	Crown diameter	Average of longest and shortest crown dimensions
	Diameter (cm)	Estimated from crown diameter[2]
	Spatial coordinates	Latitude/Longitude and UTM values
Residual snag ≥2 m tall[1] no green foliage degree of lean ≤45°	Cover class	Conifer or hardwood
	Canopy position	Dominant, codominant, intermediate, suppressed
	Height (m)	For each canopy position
	Diameter class (cm)	<10.0 cm, 10.0-24.5 cm, >24.5 cm
	Spatial coordinates	Latitude/Longitude and UTM values
Downed wood ≥2 m inside the plot degree of lean >45°	Length (m)	Inside the plot boundary
	Diameter class (cm)	<10.0 cm, 10.0-24.5 cm, >24.5 cm
	Spatial coordinates	Latitude/Longitude and UTM values at both ends of the portion of structure lying inside the plot boundary

[1] NDPE guide specifies >3 m. We used ≥2 m, which is consistent with most published literature. Potential overestimation of residual trees and snags compared to the NDPE guide was minimal because all stems <10 cm diameter, where most of the 2 to 3 m tall stems would be, were excluded from analyses.

[2] Estimated using linear regression equations proprietary to R&B Cormier and partners (R. Cormier, R&B Cormier, pers. comm., 2007).

Supporting data

In addition to collecting the residual structure data, we assembled additional information about the forest cover, site conditions, and fire variables in each of the study fires to help explain the sources of variability associated with the abundance estimates. This information was appended to the database containing the 660 sample point estimates of residual abundance.

Forest cover

As mentioned above, pre-fire forest cover information was obtained from MNR FRI (OMNR 2000), except for the fires in ecoregion 4W, for which we relied on data from Quetico Park (Quetico Forest Inventory 2000; http://ims.legacyforest.ca/main/; accessed September 2006). The forest cover type classifications were imported directly from FRI to describe preburn composition at plot level.

Site conditions

Data on the locations of water bodies and wetlands within the study fires were extracted from Ontario Base Maps. Soil moisture regime information came from the Ontario Land Inventory database (OMNR 1977). We used ArcGIS software (ESRI 2004) to calculate slope and aspect for each sample point from the elevation data, as well as the distance and direction to the nearest water body and wetland, and calculated the area and perimeter of the nearest water body and wetland from the respective maps.

Fire

The location of the fire perimeters, fire suppression activity (hose lines and fire breaks), and fire weather data were extracted from MNR fire reports and the Daily Fire Operations Support System (DFOSS) database. From the raw data, we used ArcGIS software (ESRI 2004) to calculate the distance and direction to the fire perimeter for each sample point.

We also estimated fire intensity for each of the sample points using the method described by Smith et al. (2008). The intensity of the fire at each sample point was categorized based on a ranking system that links evidence of fire damage to snag branches with a fire intensity ranking based on the Canadian Forest Fire Behaviour Prediction (FBP) system (Forestry Canada, Fire Danger Group 1992). The ranking system and criteria for determining fire intensity are outlined in Table 4.

Table 4. *Ranking system and criteria used to estimate fire intensity for each high-resolution photograph (HRP) sample point.*

Fire intensity rank	Fire intensity (kW·m^{-1})	Criteria for assigning HRP sample points to intensity rankings
1	<10	>75% of plot unburned
2	10 -500	<75% of plot unburned; snag branch structure intact
3	501-2,000	Partial (< 10%) snag fine branch destruction
4	2,001-4,000	Near complete (>10-90%) snag fine branch destruction
5	>4,000	Complete (>90%) snag fine branch destruction

Field Sampling

Monitoring temporal changes

Four of the 11 study fires were selected to establish field sample plots in which to monitor temporal changes in post-fire residual structure. These fires were located in areas where no forest harvesting (e.g., salvage logging) is likely to occur in the foreseeable future to permit long-term monitoring of residuals, as well as for logistical considerations. This resulted in selecting four fires from northwestern Ontario.

A subset of the HRP sample points within these fires were selected as candidates for field sampling. The most important criterion used to select the field plots was that each must contain residual trees so that their fate could be monitored. Among the HRP sample points that included residual trees, field plots were selected that represented a range of pre-fire forest cover types and stand ages. Among the possible sample points, 50 were selected as field sample plots based on their accessibility, with 10 plots from THU-030, 14 from THU-031, 14 from THU-067, and 12 from FOR-014. Figure 8 shows the locations of the field sample plots within the study fires.

The identical 500 m^2 circular plots that were used to interpret the HRP images were located in the field using GPS to identify the plot centres and the residual structure within the plots was visually matched with the structure on the HRP images. All residuals in these plots were tagged and enumerated within 3 months after the fire (2005) and were monitored for two more years (2006 and 2007). Figures 9 to 11 provide examples of the residual structure in these field plots. During each visit to the field plots, the mortality and fall rate of residual trees and fall rate of residual snags were enumerated. These data were used to compare the status of all stems in a plot from year to year.

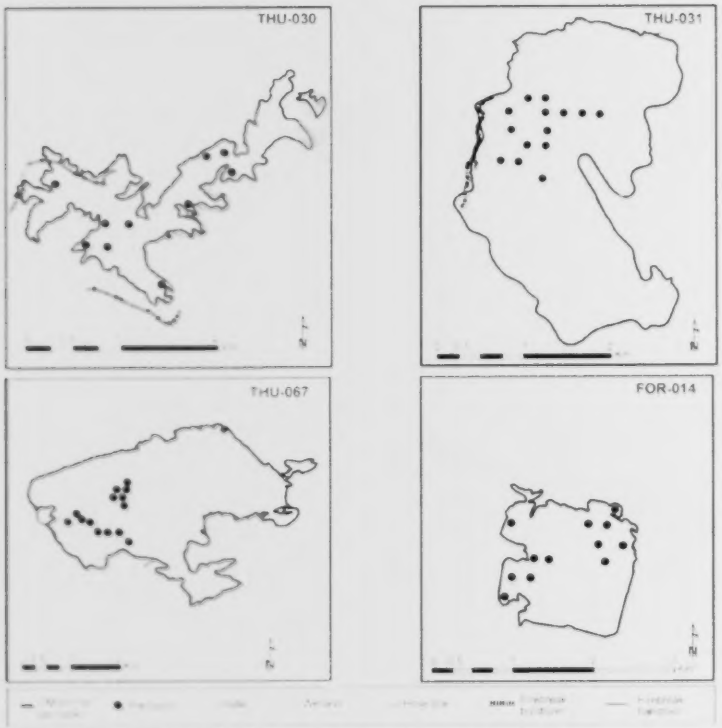


Figure 8. *Locations of the 50 field sample plots within four of the study fires.*

Figure 9. *Examples of field plots containing residual trees, snags, and downed wood during the second year after fire.*

Figure 10. *Examples of residual trees in field plots.*

Figure 11. *Examples of residual snags and downed wood in field plots.*

Error assessment

We used the same 50 field plots that were sampled to observe temporal changes in the post-fire residual structure to assess errors in interpretation of the HRP images (ground-truthing). During the 2005 sampling year, we counted all residuals within these plots based on the same definitions used to interpret the HRP information (Table 5). Field counts were guided using enlarged HRP images, and each stem tallied was spatially matched with the corresponding stem in the photos and assessed to determine its residual category, species, and diameter, as well as the length inside the plot for downed wood.

***Table 5**. Summary of post-fire residual structure categories and the variables measured in field plots used for error assessment.*

Residual category and description	Variable interpreted	Additional information
Residual tree ≥2 m tall green foliage present	Species	—
	Diameter (cm)	At breast height
Residual snag ≥2 m tall ≤45° lean no green foliage	Species	—
	Diameter (cm)	At breast height, measured along the stem from the root collar
Downed wood ≥2 m inside plot >45° lean >75% of length lying on or above the ground	Species	If not certain, then classified as conifer or hardwood
	Length (m)	Inside the plot boundary
	Diameter	At centre of portion of log within the plot boundary

These records were compared with the photo interpreted data to assess the accuracy of the estimates of residuals and determine any correction factors that should be applied to the larger data set. We expressed errors of commission (interpreter identifies a non-existent entity) as *commits*, errors of omission (interpreter misses an existing entity) as *omits*, and matching entities (interpreter correctly identifies an existing entity) as *matches*. As well, each standing residual, whether a commit, omit, or match, was classified as live (residual tree) or dead (residual snag). Residual trees could be then be correctly or incorrectly classified by species and residual snags could be correctly or incorrectly classified as deciduous or coniferous.

To determine the frequency of occurrence of each error type, we compared the field data from the 50 ground-truthed plots with the interpreted data from the HRP sample points. Conditional probabilities for each type of error were measured as rates per category of field-observed redsidual structure. We estimated these rates using the proportion of entities in a group with that error. For example, the probability of the interpreter failing to record a residual snag was calculated as the ratio of the number of snags found in the field but not by the interpreter, divided by the number found by the interpreter (fractions were rounded to the nearest integer). This approach was used to allow us to apply the resulting error models to the 610 sample points that were not ground-truthed, using the field-observed abundances as input. Because the error rates for the residual trees were derived as a function of the number of residual trees in the field plots, we adjusted the interpreted data on residual tree abundance only when at least one residual tree occurred in a sample point. This approach avoided confounding the effects of high fire intensity (which results in absence of residual trees) with the interpretation error adjustment.

A few examples of possible sources of interpretation error included: residual trees omitted when they were small and/or when their canopy was hidden under a larger tree; residual snags omitted when stem densities were very high making them difficult to count; downed wood omitted when residual densities were high; and residuals erroneously included in or excluded from a plot (Figure 12).

Figure 12. *Examples of common sources of photo interpretation errors. In some cases, standing structure was missed because it was overtopped by adjacent stems (left), or standing structure near the edge of the plot was erroneously excluded or included (right).*

We used the error analysis results to adjust the residual ab[illegible]ata (for residual trees and snags only) to reduce the bias caused by interpretation error. Linear r[illegible]ssion of the interpreted data versus the ground-truthed data for both residual trees and snags revealed a linear relationship with a high degree of predictability (Figure 13). We therefore used these regression equations (where y= field-counted entities and x =photo-interpreted entities) to adjust data in the remaining 610 sample points that had not been ground-truthed, based on their interpreted abundances of residual trees and snags. We concluded that this correction increased the accuracy of the data without introducing a new bias because deviations from the predicted values were normally distributed (results not shown).

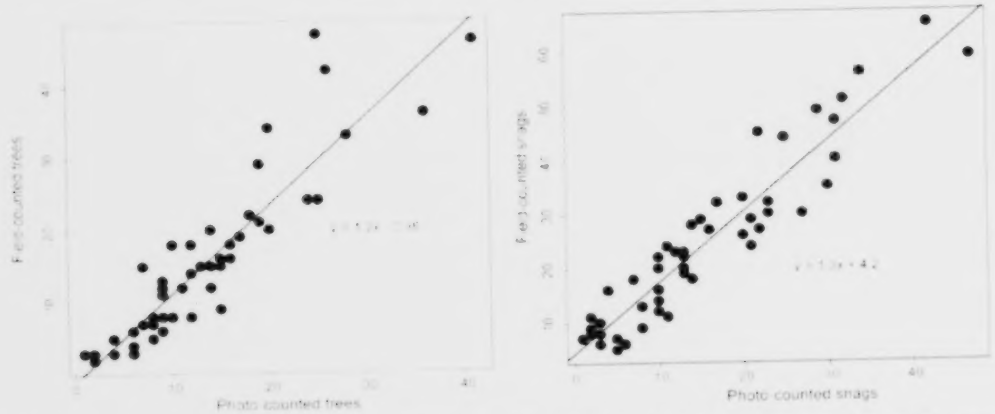


Figure 13. *Regressions of residual trees (r^2=0.82) and snags (r^2=0.89) counted in the field sample plots (field-counted) versus those interpreted from high-resolution photography (HRP) images (photo-counted) (n=50 plots).*

Results and Discussion

Error analysis

Based on a comparison of the data collected in the 50 field plots with that obtained by photo interpretation of the same 50 HRP sample points, interpretation errors (commits, omits, and misclassifications) occurred at different rates, but most (94%) of the interpreted data was correct based on the ground-truthed measurements. Detailed error analysis results are not presented here but, in general, the most common type of error was omission (30% of existing entities were missed), and this error type was more common for residual snags than for residual trees. The least common type of error was commission (6% of entities were added), and, the vast majority of these were residual trees (88%). Thus, the interpreter counted fewer residual snags than actually existed, but added some residual trees. Most residual trees were classified correctly as to species by the interpreter, and almost all residual snags were classified correctly as coniferous or deciduous. In contrast to the reasonable error rates for residual trees and snags, interpretation error rates for downed wood were high, with an average probability of a match of 42% and an average probability of an omit of 58%.

Abundance values presented for residual trees and snags represent the adjusted values for the 660 ~0.05 ha HRP sample points. For the residual trees, we used the following equation:

$y = 1.2x - 0.98$ (r^2=0.82)

Where y = corrected number of residual trees, and x = number of residual trees obtained by photo-interpretation

For residual snags, we used the following equation:

$y = 1.3x + 4.2$ (r^2=0.89)

Where y = corrected number of residual snags, and x = number of residual snags obtained by photo-interpretation

Given the high overall error rate, the downed wood error analysis was not pursued further and the downed wood abundance values obtained by photo interpretation were not adjusted to account for interpreter error.

Abundance and variability of residuals

In the following sections, the abundance and variability of residuals that occurred immediately after the fire are described for each of the three ecoregions in which fires were sampled, for each of the 11 fires sampled, and overall for each category of residuals – residual trees, residual snags, and downed wood.

In estimating the abundance of residuals, we applied three different approaches. First we used the numbers of residual trees and snags, and volume of downed wood, as per-plot (0.05 ha) values. These values are direct estimates of the abundance of residuals based on the HRP sample points, free of assumptions about their distributions.

Bootstrapping is a statistical method that uses numerical simulation to estimate unknown population parameters from sample distributions (Chernick 1999). The characteristic feature of the approach is the assembly of many "replicate" data sets by sampling with replacement from the original data. The simulated data sets are considered replicates because sampling with replacement causes the expected frequency

of any value in the simulated data set to be the same as its frequency in the original data. For each replicate data set, the parameter of interest is calculated and the distribution of this parameter over the replicated data sets can be used to estimate its true value. Bootstrapping is particularly useful for complex parameters of distributions, where confidence intervals are difficult to derive algebraically, and is used in a variety of ecological analyses (e.g., Lillegård et al. 2005, Potvin and Poff 1993).

Bootstrapping is an attractive approach because it is simple and robust with respect to the underlying frequency distribution of the data. However, it relies on the assumption that the data points are independent of one another, which affects the possibilities for ecological inference. However, where this assumption is acknowledged and considered, bootstrapping can allow tentative answers to questions that would otherwise be intractable. In summary, bootstrapped estimates are much more robust with respect to plot-to-plot variability than simply multiplying the abundance per plot (in this case by 20) to scale the results to a per-hectare level, which can lead to vast underestimates of the per-hectare abundance (given the presence of many plots with zero trees).

Second, we estimated the expected abundance of residuals per hectare by means of bootstrapping (see text box) the plot-level data to repeatedly and randomly assemble groups of plots with an equivalent area of one hectare. We used bootstrapping to estimate the distribution of initial post-fire abundances in a hectare of forest as the sum of the residual abundance in 20 study plots. The steps were as follows:

1. Sample 20 random draws with replacement from the original data (n=660).
2. Calculate the sum.
3. Repeat the above steps 10,000 times.

To estimate the abundance after three years, we also bootstrapped the temporal rates in the 50 field plots using the following additional steps between steps 1 and 2 above.

1b. Sample 20 random draws from the field temporal data (n=50). Since the sample size was small, we removed the two lowest and two highest values to prevent outliers from influencing the results.
1c. Multiply each data point from 1 with a rate selected from 1b.

The third approach, a simple multiplicative scaling (multiplying the observed plot abundances by 20) is fraught with questionable assumptions relating to the spatial scaling relationships. However, we used this approach when comparing our results to those in the literature because most residual abundance estimates reported in the literature that we examined were expressed as per-hectare values, regardless of the geographic area used or the sampling intensity. None of these studies explicitly considered the scaling implications of reporting the values on a per-hectare basis as a result of using multiplicative scaling.

Probability of expected residual abundance is a more robust and useful indicator of abundance than simple extrapolations of per plot values to per ha.

We emphasize that the scope of inference of the estimates of residual abundance varies based on the method used, and that such estimates should only be applied within that context. Thus, the comparisons of our results to those in the literature are useful only in the context of *relative* differences. Estimates of the distribution and residual abundance and variability that may be more appropriate for application in science and in resource management are included in the discussion of each category of residual below.

Residual trees

Abundance per plot

The values presented in this section are direct per-plot estimates, adjusted to account for interpreter error based on the regression analysis described in the error analysis section above.

The average abundance of residual trees (live foliage present, ≥2 m tall, >10 cm in diameter) across all study fires was 8 trees per plot immediately after the fire (Table 6). Although the observed abundance varied among ecoregions, directly comparing these values is not advisable since the sample plot numbers are unequal. Residual tree abundance also varied among the fires. The lowest average number of residual trees occurred in the largest fire (NIP-020).

Residual *trees are individual live stems that occur outside contiguous unburned patches.*

Table 6. *Abundance and variability of residual trees sampled using plots in the high-resolution sample points in the same year as the fire, by ecoregion and study fire.*

Grouping of sample points	Trees (no./plot)					
	mean	percentile			range	
		25	50	75	min	max
Ecoregion						
3W (n=508)	8	0	0	10	0	126
3E (n=102)	9	0	0	17	0	67
4W (n=50)	12	3	9	19	0	38
Fire (by ecoregion)						
NIP-020 (n=309)	4	0	0	3	0	42
THU-030 (n=32)	14	0	7	18	0	86
THU-031 (n=84)	14	0	9	18	0	126
THU-067 (n=83)	15	0	9	26	0	109
COC-010 (n=31)	8	0	0	9	0	67
TIM-005 (n=27)	12	0	1	26	0	45
TIM-007 (n=20)	6	0	0	0	0	43
TIM-019 (n=17)	9	0	6	17	0	30
CHA-018 (n=7)	9	4	10	11	0	27
FOR-014 (n=32)	10	3	9	16	0	30
FOR-024 (n=18)	15	5	12	21	0	38
Overall (n=660)	8	0	0	12	0	126

The maximum number of residual trees observed was considerably higher in some of the study fires (THU-030, THU-031, THU-067, and COC-010) relative to the others. Overall, the variability in the estimates of residual tree abundance was high both within and among fires. Since the absolute number of residual trees is a function of the pre-fire tree density, that variability is confounded with the effects of fire, and this effect should be considered when using the numbers in Table 6.

In all study fires, the distribution of the number of residual trees per plot was right-skewed: most plots had few residual trees and few plots had many residual trees. Exceptions were CHA-018 and the two Quetico fires (FOR-014 and FOR-024), where residual trees were distributed more evenly among the plots. At least 25% of the sample points in most fires, and up to 50% in some, had no residual trees (Table 6). Therefore, we examined the frequency of occurrence of plots with at least one residual tree by fire. Overall, the probability of finding residual trees at any location within a boreal fire varied among the study fires from a low of 0.35 (TIM-007 and COC-010) to 1.0 (FOR-024), with a weighted average of 0.563 (Figure 14).

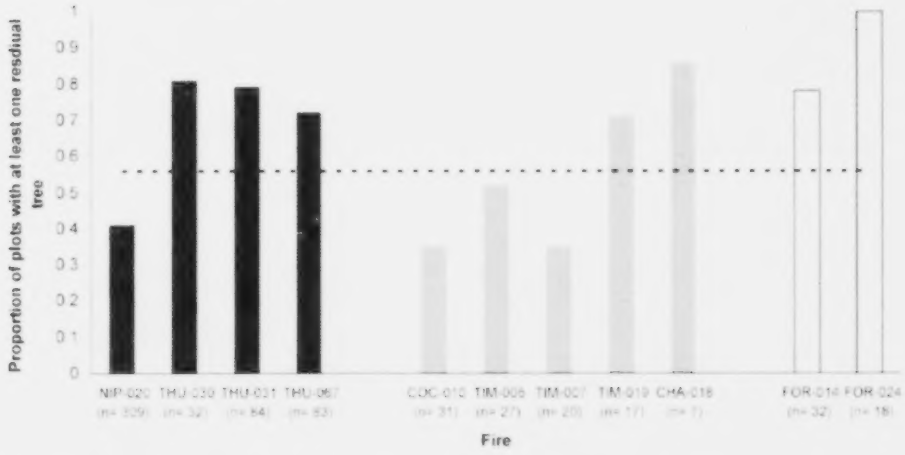


Figure 14. *Proportion of sample plots in each study fire with at least one residual tree. The dotted line indicates the weighted average value (0.563) for the 660 plots.*

Estimated distribution of abundance per hectare

In this section, we present the estimated probability distributions for per-hectare abundance of residual trees calculated using the bootstrapping approach described above. These are expressed as probabilities (i.e., relative frequency per 10,000 bootstrap simulations) of various levels of abundance per hectare. Figure 15 shows these data, both as a histogram (for which the width of the bar shows the range of abundance and the height shows the relative frequency of that range in the 10,000 bootstrap replicates) and as a cumulative probability curve, giving the relative frequency of occurrence of all values less than a specified point on the horizontal axis. The highest probability for any one range was for 100 to 150 residual trees to occur in a hectare immediately after fire (relative frequency=0.3). However, more than 150 trees per hectare were predicted more than 50% of the time, and abundance was predicted to exceed 300 trees per hectare in 10% of cases.

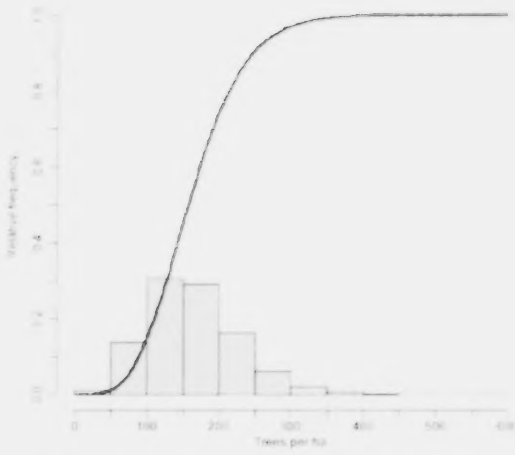


Figure 15. *Probability of estimated distribution of residual tree abundance per hectare immediately after fire shown as a point probability for ranges (histograms) and as a cumulative probability distribution (line).*

Abundance in comparison with the literature

In this section, we discuss the results of scaling our estimates of residual tree abundance to a per-hectare value by simply multiplying the plot estimates by 20 (plots were ~0.05 ha). We are aware that this simple extrapolation of per-plot data to per-hectare information is not logical when there are many zero values (i.e., no residuals in plots). However, this scaling was necessary to allow comparison of abundance estimates from this study with those reported in the literature, where all reported values are given as per-hectare values (regardless of the sampling method or plot size). Given the inappropriateness of this simple up-scaling, we remind the reader that these per-hectare values should not be used for any purposes other than simple comparisons among studies.

This scaling is for comparing with literature only.

Our estimates of the residual tree abundance in the year of fire varied widely among and within fires. As shown in Figure 16, the mean numbers of residual trees in the 11 fires ranged from 78 to 316 per hectare. This range is comparable to previously reported estimates, although the latter are estimates obtained one to five years after fire. Note also that the definition of residual trees varied among studies (Perera et al. 2007).

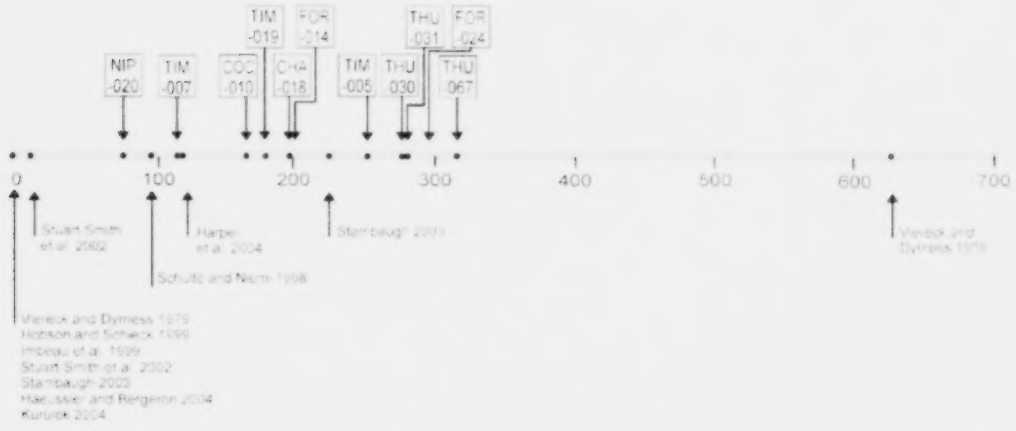


Figure 16. *Mean numbers of residual trees per hectare after boreal forest fires, sampled immediately post-fire in this study, compared with values in the literature reported for one to five years after fire.*

Residual snags

Residual snags are standing dead stems created by the fire.

Abundance per plot

The values presented in this section are direct per-plot estimates, adjusted to account for interpreter error based on the regression analysis described in the error analysis section above.

The average abundance of residual snags (no live foliage present, ≥2 m tall, <45° lean, >10 cm diameter) per plot across all study fires ranged from none to 169 immediately after the fire (Table 7). The highest mean number of residual snags per plot occurred in a mid-sized fire (THU-031).

We assume that the study fires vapourized and felled only a small fraction of snags. Thus, residual snag (i.e., trees that died as a result of the fire) abundance reflects the pre-fire stand density. Using information collected from the 50 field plots, we determined that approximately 20% of the snags were of pre-fire origin, with the remainder resulting from the fire. However, the numbers in Table 7 reflect all residual snags identified by photo interpretation in the sample plots (i.e., including both pre-fire and fire-caused snags, which are confounded).

The average abundance of residual snags varied among the fires. The lowest average number of snags occurred in the two Quetico fires (FOR-014 and FOR-024). The maximum numbers of residual snags observed in some of the study fires (THU-031, NIP-020) were several times higher than those observed in the Quetico fires. Overall, the variability in estimates of residual snag abundance was high both within and among fires.

Table 7. *Abundance and variability of residual snags sampled using plots in high-resolution sample points in the same year as the fire, by ecoregion and study fire.*

Grouping of sample points	Residual snags (no./plot)					
	mean	percentile			range	
		25	50	75	min	max
Ecoregion						
3W (n=508)	35	19	30	49	0	169
3E (n=102)	29	17	25	41	0	86
4W (n=50)	14	7	12	21	0	36
Fire (by ecoregion)						
NIP-020 (n=309)	39	22	34	51	0	139
THU-030 (n=32)	36	19	29	55	7	84
THU-031 (n=84)	26	12	18	31	0	169
THU-067 (n=83)	31	16	25	39	6	110
COC-010 (n=31)	31	10	22	47	0	85
TIM-005 (n=27)	31	23	30	41	9	56
TIM-007 (n=20)	30	21	26	37	8	64
TIM-019 (n=17)	28	9	19	30	0	86
CHA-018 (n=7)	16	9	17	25	0	32
FOR-014 (n=32)	15	8	12	21	0	36
FOR-024 (n=18)	12	6	12	17	0	26
Overall (n=660)	33	16	28	45	0	169

Estimated distribution of abundance per hectare

In this section, we present the per-plot estimates of residual snags extrapolated to a per-hectare basis using the bootstrapping approach described above. As with the residual trees, these are expressed as the probability of certain abundance ranges in a given hectare. Figure 17 shows this estimated probability distribution, both as a histogram and using a cumulative probability curve. The highest probability for any one range of abundance is for 600 to 700 snags to occur per hectare immediately after fire (relative frequency=0.4). More than 700 snags per hectare were observed 50% of the time, but only rarely were more than 1000 snags per hectare observed.

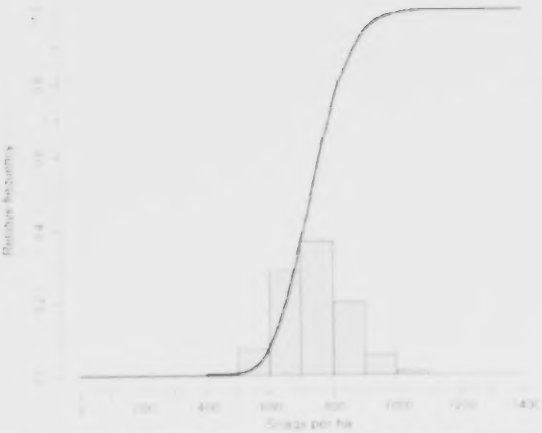


Figure 17. *Probability of estimated distribution of snag abundance per hectare immediately after fire shown as a point probability for ranges (histograms) and as a cumulative probability distribution (line).*

Abundance in comparison with the literature

As was the case for residual trees, we compared the abundance estimates from this study with those reported in literature (where values are generally reported as per-hectare values regardless of sampling method or plot size) by scaling our estimates of residual snag abundance to per hectare by simply multiplying the per-plot estimates by 20 (since sample plots were ~0.05 ha). We remind the reader that these per-hectare values should not be used for purposes other than simple comparisons among studies.

This scaling is for comparing with literature only.

Our estimates of the abundance of residual snags in the year of fire varied widely among and within fires. As shown in Figure 18, the mean number of snags in the 11 fires ranged from 238 to 792 per hectare. These values were comparable with previously reported estimates in the published literature, although the latter were estimated one to five years after fire. Note also that the definition of residual snags varied among studies (Perera et al. 2007).

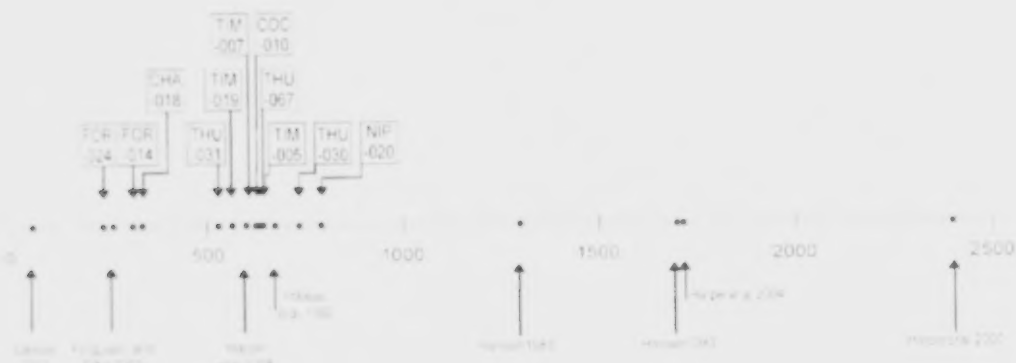


Figure 18. *Mean numbers of residual snags per hectare in boreal forest fires sampled in this study compared to values provided in the literature for one to five years post-fire.*

Downed wood

The estimate volumes of downed wood provided in this section should be considered in the context of the high error rates associated with the photo interpretation of downed wood. (As discussed above, downed wood abundance was underestimated in the HRP interpretation.)

Abundance per plot

The values presented in this section are the direct per-plot estimates, not adjusted for interpreter error. The average volume of downed wood (≥2 m length inside sample plot, >45° lean, >10 cm diameter at centre, >75% of length above the ground) across all study fires ranged from 0 to 13.64 m^3 per plot (Table 8).

As was the case with snags, the abundance of downed wood estimated based on photo interpretation after the fire is confounded with downed wood already there before the fire. Using information collected from the field plots, we determined that about 68% of the downed wood was of pre-fire origin, with the remainder resulting from the fire. However, the numbers in Table 8 reflect all downed wood identified by photo interpretation in the sample plots (i.e., including both pre-fire and fire-caused downed wood, which are confounded).

Table 8. *Abundance and variability in the volume of downed wood sampled using plots in high-resolution photography sample points in the same year as the fire, by ecoregion and study fire.*

Grouping of sample points	Downed wood volume (m^3/plot)					
	mean	percentile			range	
		25	50	75	min	max
Ecoregion						
3W (n=508)	1.77	0.82	1.31	2.20	0.03	12.13
3E (n=102)	1.74	0.22	0.61	2.02	0.00	13.64
4W (n=50)	0.99	0.31	0.83	1.51	0.03	3.60
Fire (by ecoregion)						
NIP-020 (n=309)	1.70	0.78	1.25	2.05	0.03	12.13
THU-030 (n=32)	2.05	0.92	1.33	2.42	0.34	9.91
THU-031 (n=84)	1.66	0.77	1.34	2.30	0.11	6.93
THU-067 (n=83)	2.05	0.98	1.40	2.51	0.05	9.92
COC-010 (n=31)	0.74	0.13	0.29	0.63	0.01	4.99
TIM-005 (n=27)	4.00	0.96	3.20	5.22	0.07	13.64
TIM-007 (n=20)	1.36	0.33	1.18	2.05	0.02	5.48
TIM-019 (n=17)	0.68	0.04	0.49	0.81	0.00	2.85
CHA-018 (n=7)	1.04	0.36	0.78	1.65	0.00	2.47
FOR-014 (n=32)	1.03	0.24	0.98	1.54	0.03	3.01
FOR-024 (n=18)	0.93	0.34	0.60	1.33	0.05	3.60
Overall (n=660)	1.71	0.65	1.21	2.11	0.00	13.64

Estimated distribution of abundance per hectare

In this section, we present the per-plot estimates of downed wood volume extrapolated to a per-hectare basis using the bootstrapping approach described above. As was the case with the residual trees, these are expressed as an estimated probability (relative frequency per 10,000 bootstrap simulations) of expected per-hectare abundance values.

Figure 19 shows this probability distribution both as a histogram (showing the relative frequency of various ranges of abundance) and a cumulative probability curve (showing the probability that abundance will be less than a specified value on the horizontal axis). The highest probability for a single range was for 30 to 35 m^3 of downed wood per hectare to occur immediately after fire (relative frequency=0.3), but the volume of downed wood occasionally exceeded 60 m^3 per hectare.

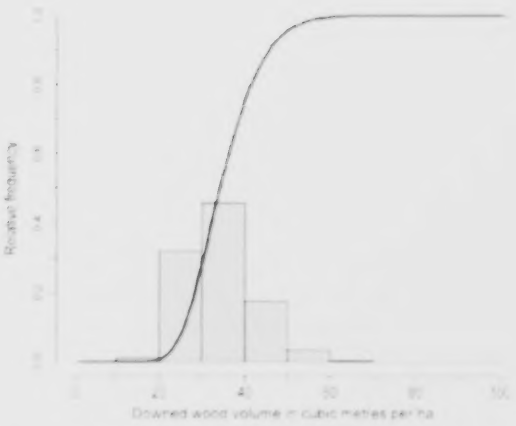


Figure 19. *Probability of estimated distribution of downed wood volume per hectare immediately after fire shown as a point probability for ranges (histograms) and as a cumulative probability distribution (line).*

Abundance in comparison with the literature

We did not compare our results with those reported in the literature because the metrics used to measure downed wood and to present results varied too widely in the literature for the results of such a comparison to be meaningful. For details, see Perera et al. (2007).

Sources of variability for residual abundance

In this section, we investigate possible sources of variability for residual abundance (as reported in literature and reviewed by Perera et al. 2007). These sources of variability include pre-fire forest cover characteristics – cover type, stand age, and diameter class; site factors – slope, aspect, soil moisture, distance to water, and distance to a wetland; and fire variables – geometry and intensity. Results are presented separately for each source of variability for each category of residual – residual trees, residual snags, and downed wood – and then compared with results reported in the literature.

Values represent the direct per-plot abundances presented in relation to each site factor separately, without examining interactions. More rigorous statistical testing of these associations is reported by Perera et al. (in review-a).

Results throughout this section are illustrated using box plots in which the vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable.

Residual trees

Forest cover

The species that is dominant before the fire may affect the abundance of residual trees. The median numbers of residual trees were higher in mixed conifer and hardwood stands and red and white pine-dominated stands, but these were also the cover types with fewer sample points, and variability is moderate to high for all cover types. The highest variability in the number of residual trees occurred in the black spruce and jack pine cover types, but these also had the most sample points (Figure 20a).

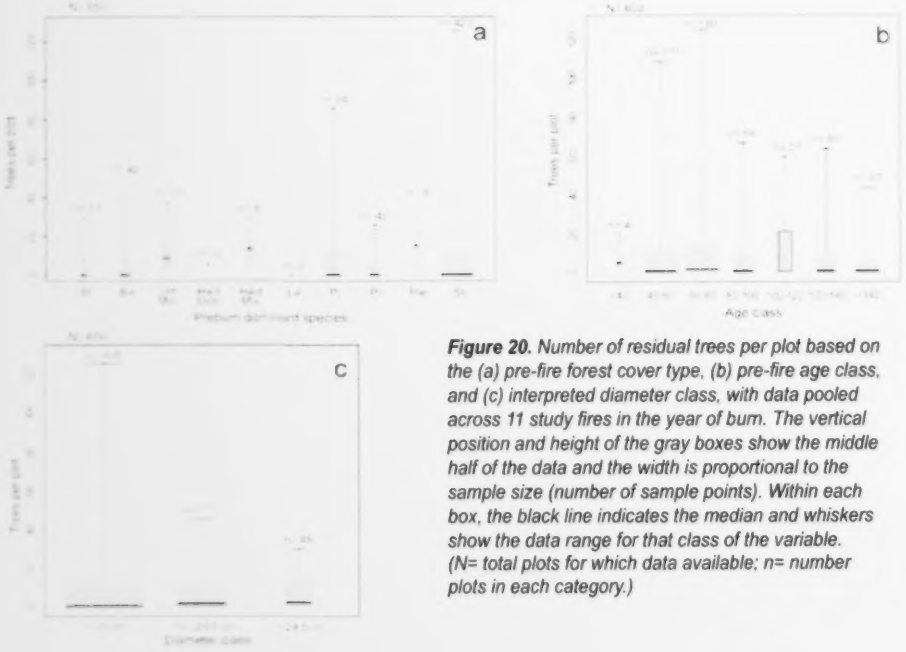


Figure 20. *Number of residual trees per plot based on the (a) pre-fire forest cover type, (b) pre-fire age class, and (c) interpreted diameter class, with data pooled across 11 study fires in the year of burn. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Stand age did not seem to affect the number of residual trees in the year of the fire, but variability within age classes was high (Figure 20b). The variability in the number of residual trees was highest for the smallest diameter class (Figure 20c).

We found no generalized associations between residual tree abundance and characteristics of pre-fire cover types (species, age) or diameter class of residuals at the sample plot level (Perera et al. in review-a). However, many associations with occurrence and abundance of residual trees have been reported in the literature for species and cover type (Lutz 1956, Scotter 1972, Apfelbaum and Haney 1986, Hobson and Schieck 1999, Morissette et al. 2002) as well as for tree diameter (Ohman and Grigal 1979, Kafka et al. 2001, Hely et al. 2003, Beverley and Martell 2003, Hauessler and Bergeron 2004). Many reasons are possible for our failure to detect statistically significant associations with pre-fire forest cover. For example, most of our sample points were dominated by black spruce prior to fire, thereby reducing our ability to detect differences for other forest cover types. As well, the pre-fire cover type information – for species, age, and diameter, which was extracted from FRI data – may not be accurate at our sampling resolution. Finally, reliable and simple linear associations between the occurrence of residuals and pre-fire forest cover may not exist. Instead these relationships may differ depending on the situation (thus explaining the diversity of results reported in the literature), and may involve complex non-linear interactions with other site variables.

Site conditions

As shown in Figure 21, no major associations were apparent between the number of residual trees and the site factors that we considered. No effects of soil moisture on the number of residual trees were discernable, but the variability appeared to be lower on moist sites than on dry or mesic sites (Figure 21a). No trends were evident for slope but variability appeared to decrease as slope increased (Figure 21b). However, this effect may be a function of sample size. Variability in the number of residual trees was lowest on N–NE and SW–W aspects (Figure 21c). Variability in the number of residual trees was lowest closer to water than at distances >100 m (Figure 21d), and increased with increasing distance from the nearest wetland, with lower variability at distances <500 m (Figure 21e).

Our results showed no significant associations between the number of post-fire residual trees and site conditions (soil moisture, slope, aspect, and the distance to water or wetland) (Perera et al. in review-a). However, as Perera et al. 2007 detailed, the literature indicates some associations between site factors and the occurrence of residual trees, with higher occurrence noted in wetter areas (Lutz 1956, Scotter 1972, Nordin and Grigal 1975, Apfelbaum and Haney 1981, 1986), in spruce bogs (Scotter 1972, Viereck and Dyrness 1979, Apfelbaum and Haney 1981), and at the margins of peatlands (Arsenault 2001). The latter result may be from observations of single or few occurrences rather than trends that can be generalized across boreal fires.

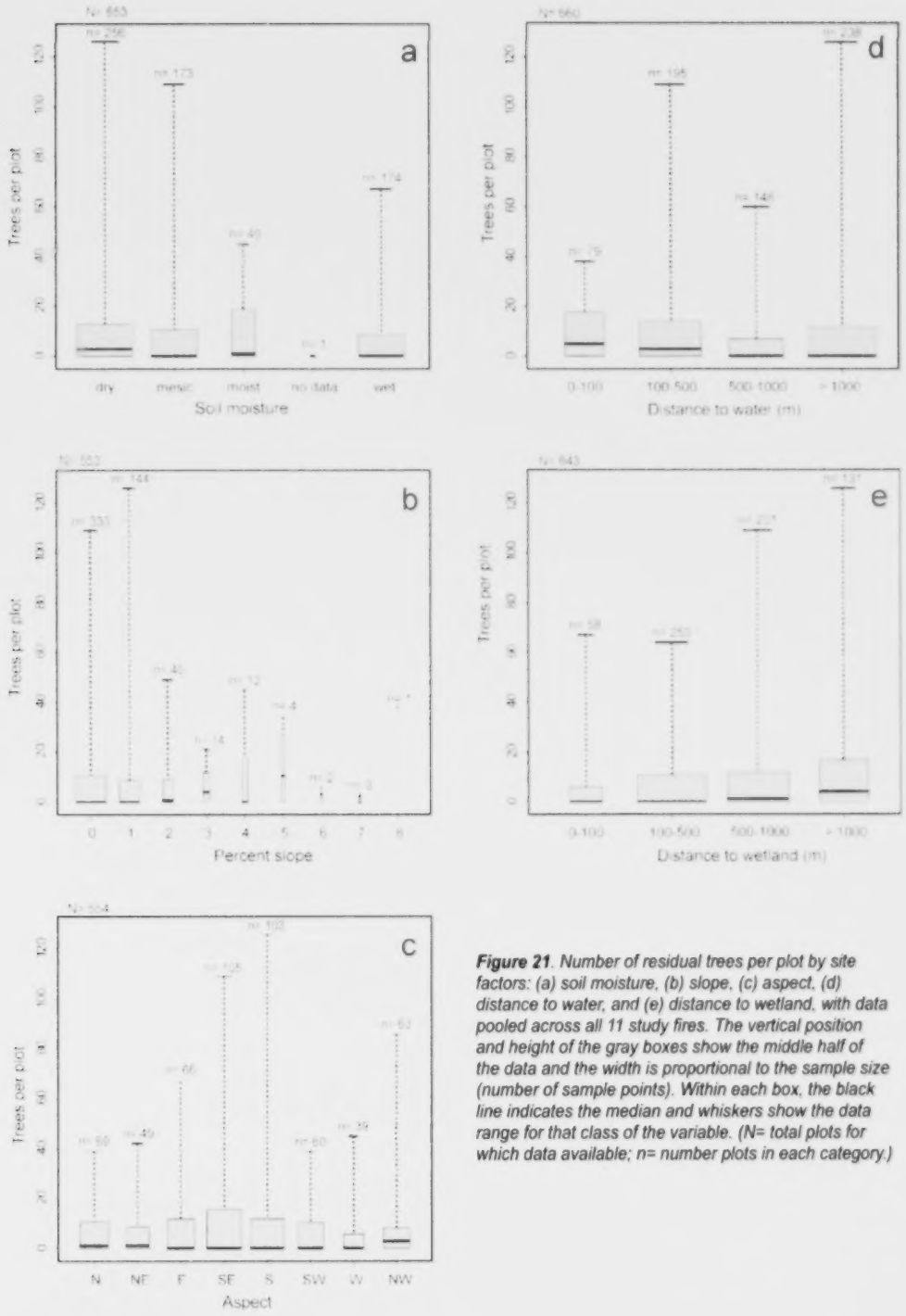


***Figure 21**. Number of residual trees per plot by site factors: (a) soil moisture, (b) slope, (c) aspect, (d) distance to water, and (e) distance to wetland, with data pooled across all 11 study fires. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Fire intensity

Residuals are believed to be associated with several characteristics of forest fires, such as fire size and the location within fire events. However, these are all surrogate measures of the effects of fire intensity. In our study, we examined the associations of residuals with indirect measures of fire intensity (i.e., fire size and geometry) expressed using several descriptors.

Fire size did not appear to determine the number of residual trees immediately after fire (Figure 22).

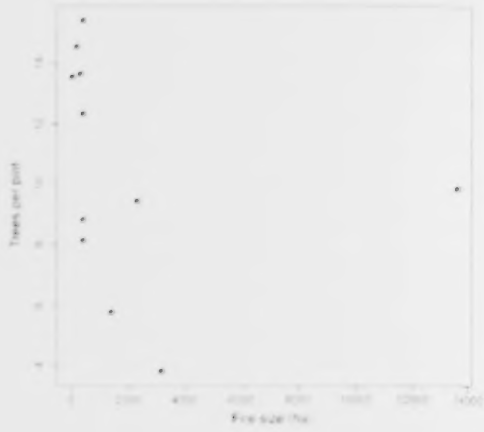


***Figure 22**. Average number of residual trees per plot in the year of fire in relation to fire size (N=11 fires).*

We examined the distance of each sample point from the fire's perimeter and categorized the sample points into distance classes of 0-100, 100-500, 500-1,000 and >1,000 m. Although the median number of residual trees per plot decreased with increasing distance to the fire perimeter, the variability was high within each distance class, and generally decreased with increasing distance from fire perimeter (Figure 23).

Fire geometry was further assessed using the distance and direction from each sample point to the nearest point on the fire perimeter. We determined the fire perimeter using MNR fire reports, but small sections of the perimeter were occasionally excluded to prevent anthropogenic disturbance, such as fire suppression or logging, from influencing the results. The direction to the nearest point on the perimeter was measured using the horizontal and vertical distances, respectively.

Patterns in the variability at various distances from the fire perimeter indicated a possible trend towards higher variability and higher residual tree abundance closer to fire edges (Figure 24). This is one of the stronger trends, and it echoes trends seen in the previous results (Figure 23). Distance to fire perimeter was thus a significant factor in whether or not residual trees would be found at a site, but this effect varied among fires. For most fires, the probability of encountering at least one residual tree was predicted to decline with distance from the perimeter (Figure 24), with the exception of THU-067. However, because our analysis fitted average conditions for highly variable data and was intended only to explore trends, we do not suggest using it to predict residual tree occurrence in general. Nonetheless, the trends indicated by

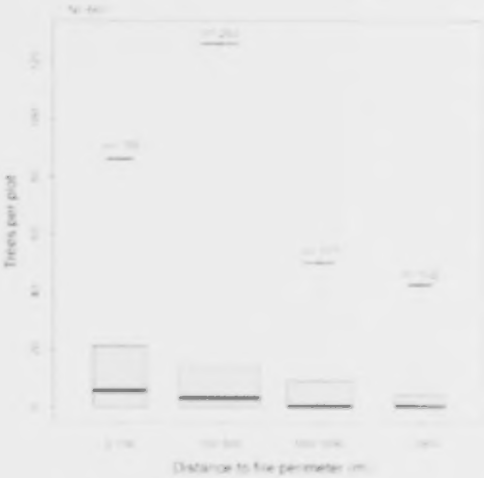


***Figure 23**. Number of residual trees per plot by distance to fire perimeter, with data pooled across all 11 study fires. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

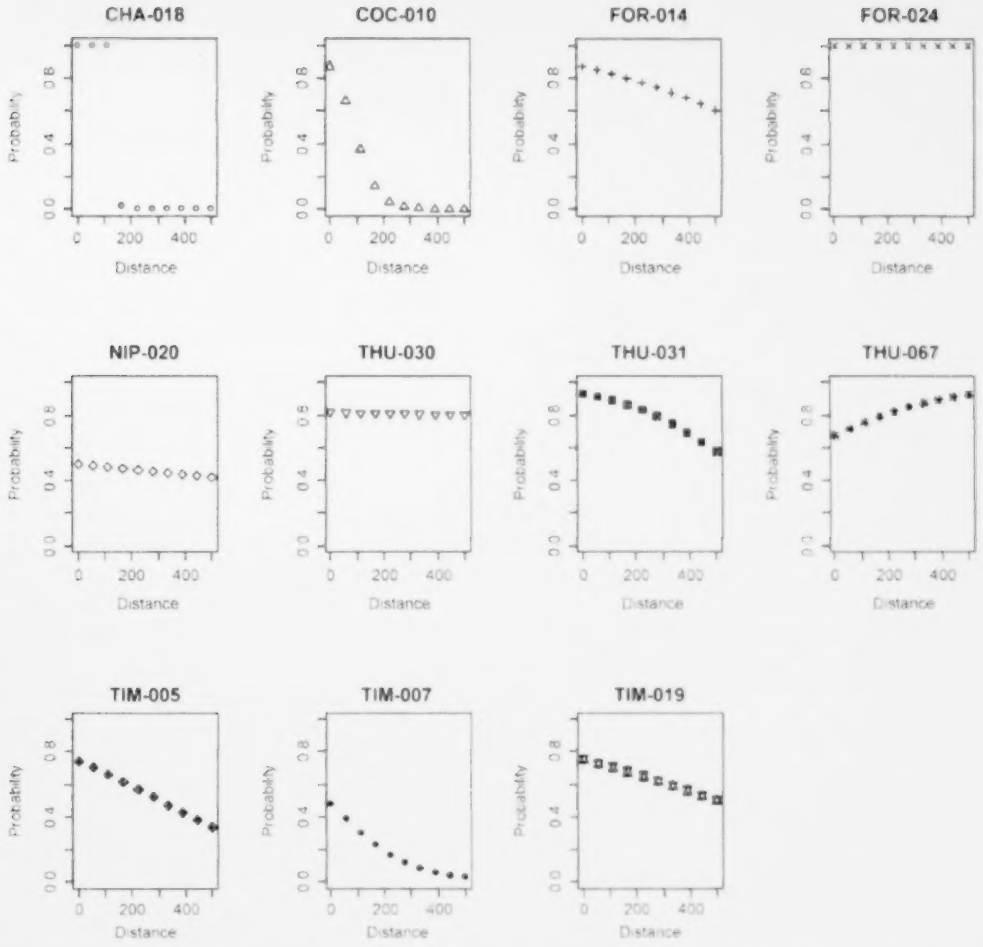


Figure 24. *Predicted effect of distance to the fire perimeter (m) on the probability of at least one residual tree remaining after the fire for each of 11 study fires.*

our data agree with those in the literature, where a higher probability of residual tree occurrence is reported nearer to a fire's perimeter (Lutz 1956, Methven et al. 1975, Haeussler and Bergeron 2004, Harper et al. 2004).

In addition to the surrogate estimates, we examined the effects of fire intensity on residual tree abundance directly, using the fire intensity categories developed by Smith et al (2008). In this analysis, we found that the number of residual trees decreased with increasing fire intensity ranking with very few trees surviving at fire intensity ranks 4 (2,000-4,000 $kW \cdot m^{-1}$) and 5 (>4,000 $kW \cdot m^{-1}$) (Figure 25). Variability in the number of residual trees was much higher for lower fire intensity ranks. This trend was also evident in the literature, with many reports of high residual tree abundance where fire intensity is low (Lutz 1956, Methven et al. 1975, Morissette et al. 2002, Hely et al. 2003, Haeussler and Bergeron 2004, Harper et al. 2004).

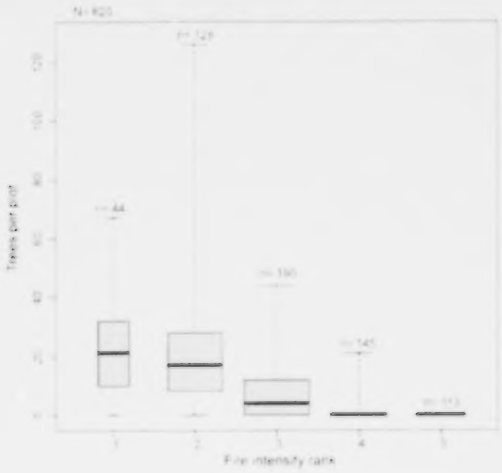


Figure 25. *Number of residual trees per plot by fire intensity rank, with data pooled across all 11 study fires. Fire intensity ranks are based on the Canadian Forest Fire Behaviour Prediction System, where 1=< 10, 2=10-500, 3=500-2,000, 4=2,000-4,000, 5>4,000 kW·m^{-1}. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Residual snags

Forest cover

As shown in Figure 26, no major associations were apparent between the number of residual snags and the site factors that we considered. The number of snags per plot was highly variable for all pre-fire forest cover types, and especially for black spruce, but this cover type also had the most sample points (Figure 26a). The lowest numbers of snags and least variability occurred in the hardwood-dominated, mixedwood, and red and white pine forest cover types, but these also had the fewest sample points. Few snags occurred in young stands, but apart from this age class, the number of snags was highly variable within age classes (Figure 26b). Variability in the number of snags was highest for the smallest diameter class (Figure 26c).

Previous studies reported that the abundance of snags is a direct result of the pre-fire stand density (Lee and Crites 1999, Graf et al. 2000, Sander 2003), but we found only one report of an association of forest cover type with snag abundance (Miyanishi and Johnson 2002) and one report of stand age influencing snag abundance (Apfelbaum and Haney 1986) after fire.

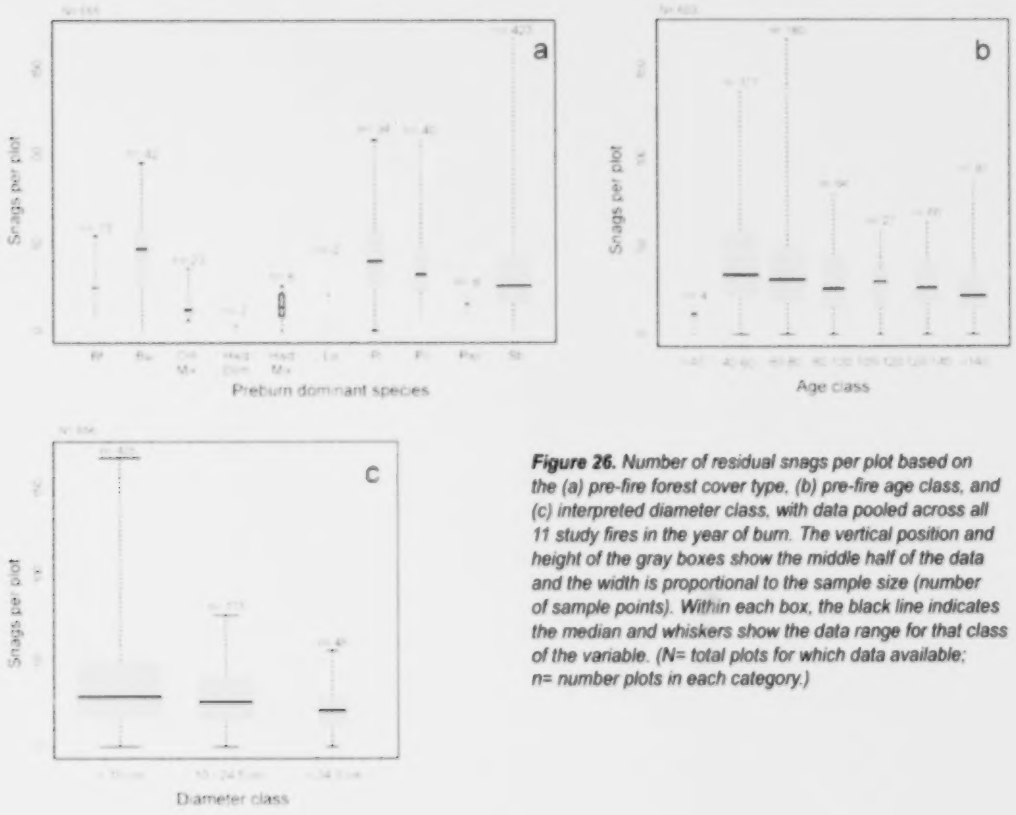


Figure 26. *Number of residual snags per plot based on the (a) pre-fire forest cover type, (b) pre-fire age class, and (c) interpreted diameter class, with data pooled across all 11 study fires in the year of burn. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Site conditions

As shown in Figure 27, no associations were evident between the number of residual snags and any of the site factors that we considered. However, the variability in snag abundance was least on moist and mesic sites (Figure 27a). As well, variability in snag abundance appeared to decrease as slope increased but this is likely the result of diminishing sample sizes rather than an actual trend (Figure 27b). Variability in snag abundance was highest for south-facing aspects (Figure 27c) and at a distance of more than 1,000 m from water (Figure 27d) or from the nearest wetland (Figure 27e).

Some reports in the literature have associated the abundance of residual snags with site conditions, and specifically with soil moisture, but they were inconsistent (Apfelbaum and Haney 1986, Lee and Crites 1999, Bond-Lamberty et al. 2003, LeGoff and Sirois 2004, Harper et al. 2005). No associations of snag occurrence with slope, aspect, or distance to water or wetlands were reported in the literature.

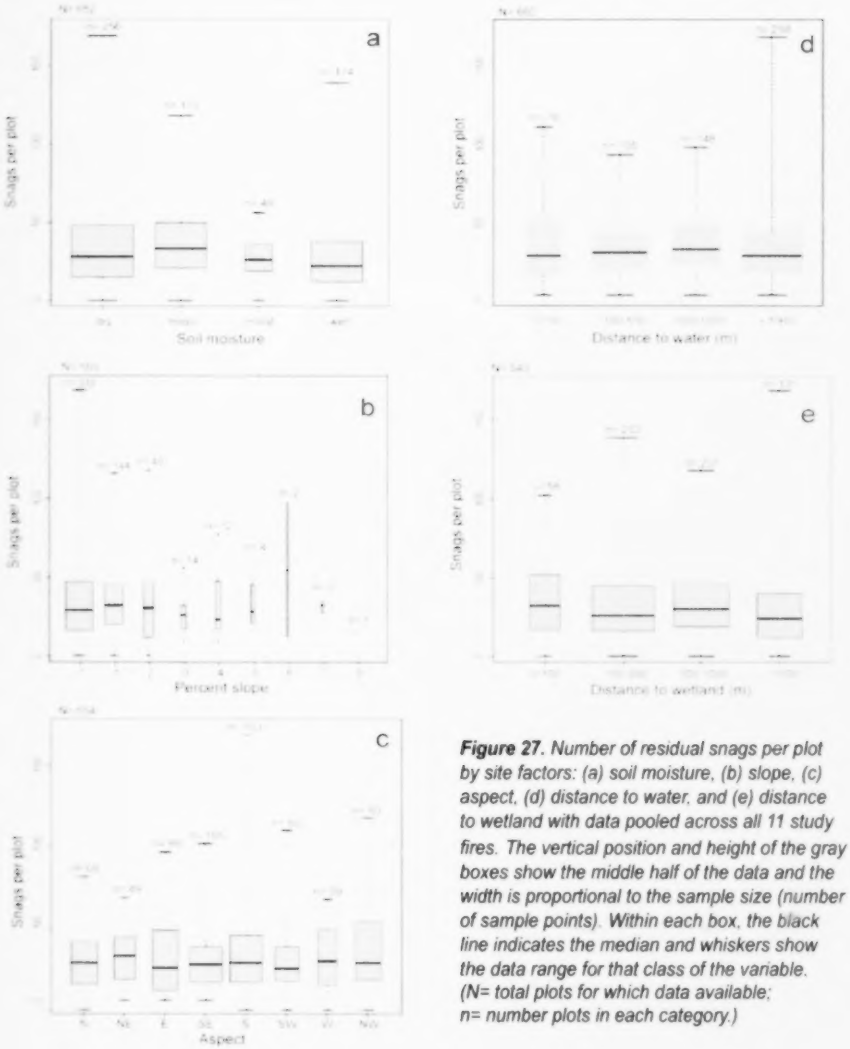


Figure 27. *Number of residual snags per plot by site factors: (a) soil moisture, (b) slope, (c) aspect, (d) distance to water, and (e) distance to wetland with data pooled across all 11 study fires. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Fire intensity

As was the case with residual trees, we examined the effect of fire on residual snags as a function of fire size and geometry as well as through direct estimates of fire intensity. Fire size did not affect the number of residual snags remaining after fire in our study fires (Figure 28).

No effect of distance to the fire perimeter on the number of residual snags was discernable in our data, but the variability was high within the distance classes (Figure 29a). However, we found that the fire intensity ranks (Smith et al. 2008) were associated with the abundance of snags in our study fires: the mean number of snags appeared to increase as fire intensity ranks increased (Figure 29b). Variability in the number of snags was lowest at the lowest fire intensity rank. This agrees with reports in the literature that fire intensity may be important in explaining the variability in post-fire snag abundance in boreal forests (Apfelbaum and Haney 1986, Sander 2003).

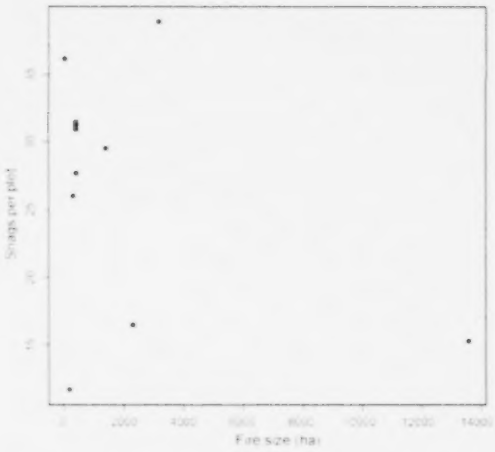


***Figure 28**. Average number of residual snags per plot in the year of fire in relation to fire size (N=11 fires).*

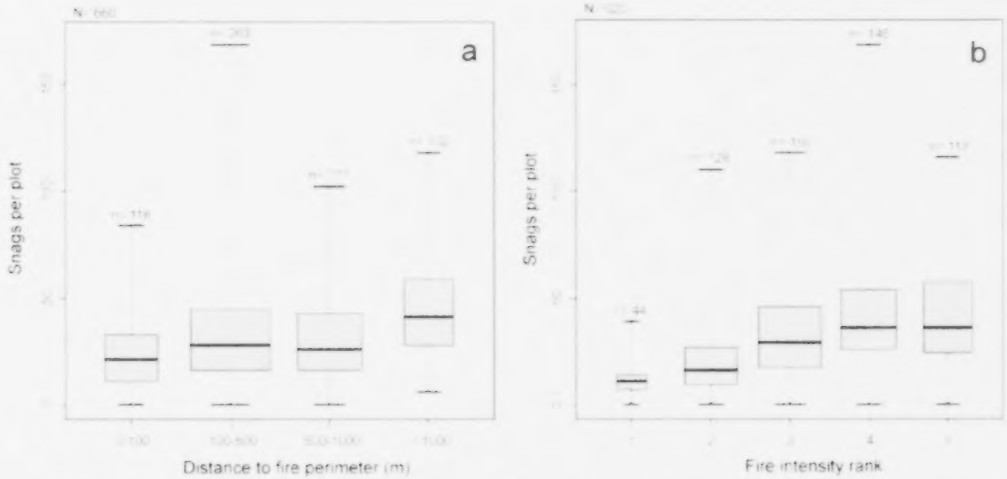


***Figure 29.** Number of residual snags per plot by (a) distance to fire perimeter and (b) fire intensity rank, with data pooled across all 11 study fires. Fire intensity ranks are based on the Canadian Forest Fire Behaviour Prediction System, where 1=< 10, 2=10-500, 3=500-2,000, 4=2,000-4,000, 5>4,000 kW·m⁻¹. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Downed wood

The estimates of downed wood volume provided in this section should be considered in the context of the high error rates associated with the photo interpretation of downed wood. (As discussed in the error analysis section, downed wood abundance was underestimated in the HRP interpretation.)

Forest cover

Our results indicate no associations between the post-fire volume of downed wood with pre-fire forest cover type, but variability was lower for some cover types than for others, with especially high variability in the jack pine, poplar, and black spruce cover types (Figure 30a). This could be a reflection of the differences in sample size among cover types. The lowest variability in downed wood volume occurred in the youngest age class (Figure 30b) and in the smallest diameter class (Figure 30c). In the literature, only one report was found of a possible association of downed wood volume with forest cover types (Morissette et al. 2002).

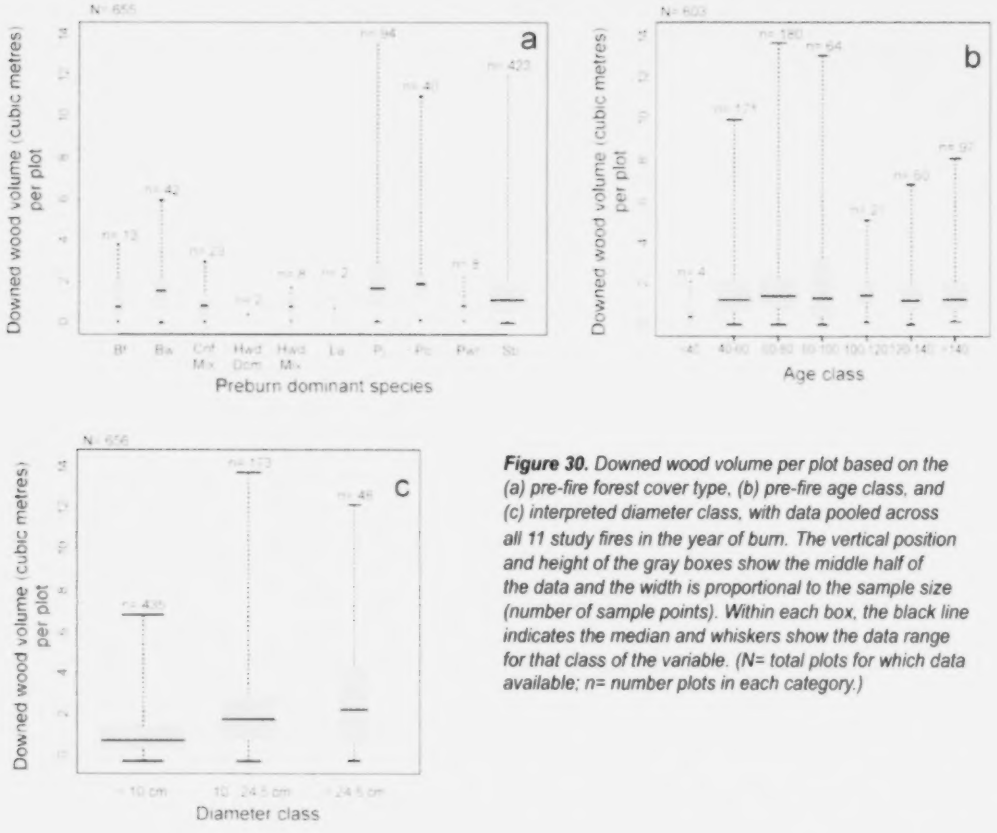


Figure 30. *Downed wood volume per plot based on the (a) pre-fire forest cover type, (b) pre-fire age class, and (c) interpreted diameter class, with data pooled across all 11 study fires in the year of burn. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Site conditions

As shown in Figure 31, the post-fire volume of downed wood does not appear to be associated with any of the site factors that we considered, and the variability in downed wood abundance was high for all site factors.

The variability in downed wood volume was lowest for wet areas and highest for moist and dry areas (Figure 31a). This variability appeared to decrease as slope increased, but this was likely due to diminishing sample sizes (Figure 31b). The NE, SW, and W aspects appeared to have the least variability in downed wood abundance (Figure 31c). Variability in downed wood volume was lower within 100 m of water than at greater distances (Figure 31d) but was not affected by the distance from wetlands, for which variability was high in all distance categories (Figure 31e). In the literature, two reports of associations of downed wood volume with site factors were found relating to soil moisture (Bond-Lamberty et al. 2003, Harper et al. 2005).

Figure 31. *Downed wood volume per plot by site factors: (a) soil moisture, (b) slope, (c) aspect, (d) distance to water, and (e) distance to wetland, with data pooled across all 11 study fires. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Fire intensity

Only the effects of distance to fire perimeter and fire intensity ranking were assessed for post-fire volume of downed wood. The distance to the fire perimeter did not appear to influence the total volume of downed wood in a plot, but variability was high for all distance categories assessed (Figure 32a). Only one report was found in the literature of the effect of fire geometry on downed wood abundance (Harper et al. 2004).

Variability in downed wood volume was least at the lowest fire intensity rank, but was high overall (Figure 32b). No reports of associations of volume of downed wood with fire intensity were found in the literature.

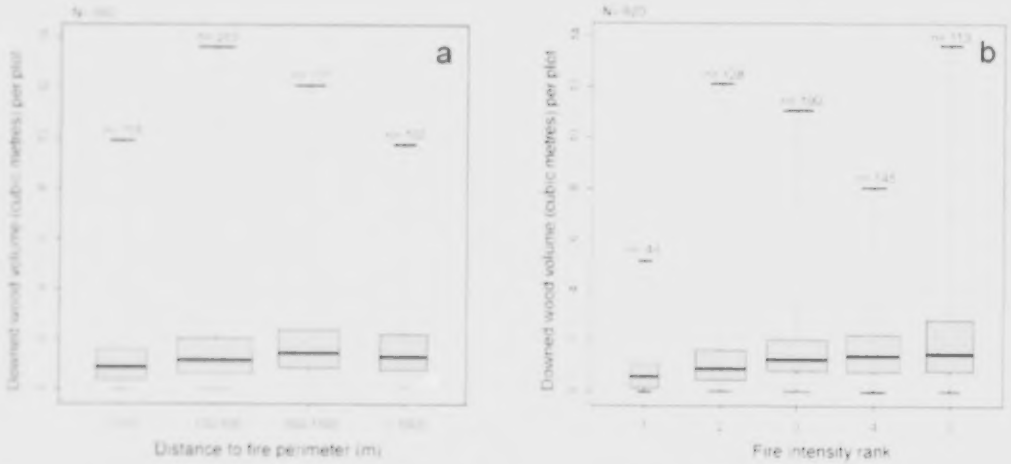


Figure 32. *Downed wood volume per plot by (a) distance to fire perimeter and (b) fire intensity rank, with data pooled across all 11 study fires. Fire intensity ranks are based on the Canadian Forest Fire Behaviour Prediction System, where 1=<10, 2=10-500, 3=500-2,000, 4=2,000-4,000, 5>4,000 kW m⁻¹. The vertical position and height of the gray boxes show the middle half of the data and the width is proportional to the sample size (number of sample points). Within each box, the black line indicates the median and whiskers show the data range for that class of the variable. (N= total plots for which data available; n= number plots in each category.)*

Longevity of residuals

We monitored the changes in the abundance of post-fire residuals for two years after the year of fire, using a subset of the 660 HRP sample points comprising 50 field plots from four of the study fires. Selection of the field plots was biased to those that included residual trees. The changes documented were mortality and fall of residual trees and fall of snags. In this section, we report the results of this monitoring, first as changes in overall abundance, followed by specific fates of residual trees and residual snags.

Overall changes in residual abundance with time

Table 9 summarizes the observed changes in the abundance of residuals over three years for each category of residual. The average abundance of residual trees decreased rapidly for all study fires, by more than 50% in each consecutive year, reaching an average of 6 per plot by the third year. Although different rates of change were observed in different fires, the abundance of residual trees in the third year was similar among fires. Observed residual snag abundance remained consistent over the three years, with only an overall 10% decrease in abundance.

Table 9. *Per-plot abundance of residual trees and snags observed over three years.*

Residual category	Study Fire (no. of sample plots)	Per-plot abundance (no.)		
		Immediately post-fire (2005)	2nd year (2006)	3rd year (2007)
Residual trees	THU-030 (n=10)	28	21	8
	THU-031 (n=14)	29	6	2
	THU-067 (n=14)	33	18	8
	FOR-014 (n=12)	20	8	6
	Mean (N=50)	**28**	**13**	**6**
Residual snags	THU-030 (n=10)	90	89	86
	THU-031 (n=14)	49	49	46
	THU-067 (n=14)	27	23	20
	FOR-014 (n=12)	42	38	35
	Mean (N=50)	**49**	**47**	**44**

Temporal changes in the abundance of post-fire residuals varied among fires and among plots within a fire (Figure 33). However, some trends were consistent across plots and among fires: (a) most of the residual tree and snag fall occurred between the year of the fire and the second year; (b) many of the residual trees became snags between the year of the fire and the second year, and this tree mortality continued until the third year; and (c) by the third year, almost all plots contained mostly snags and downed wood, with few residual trees.

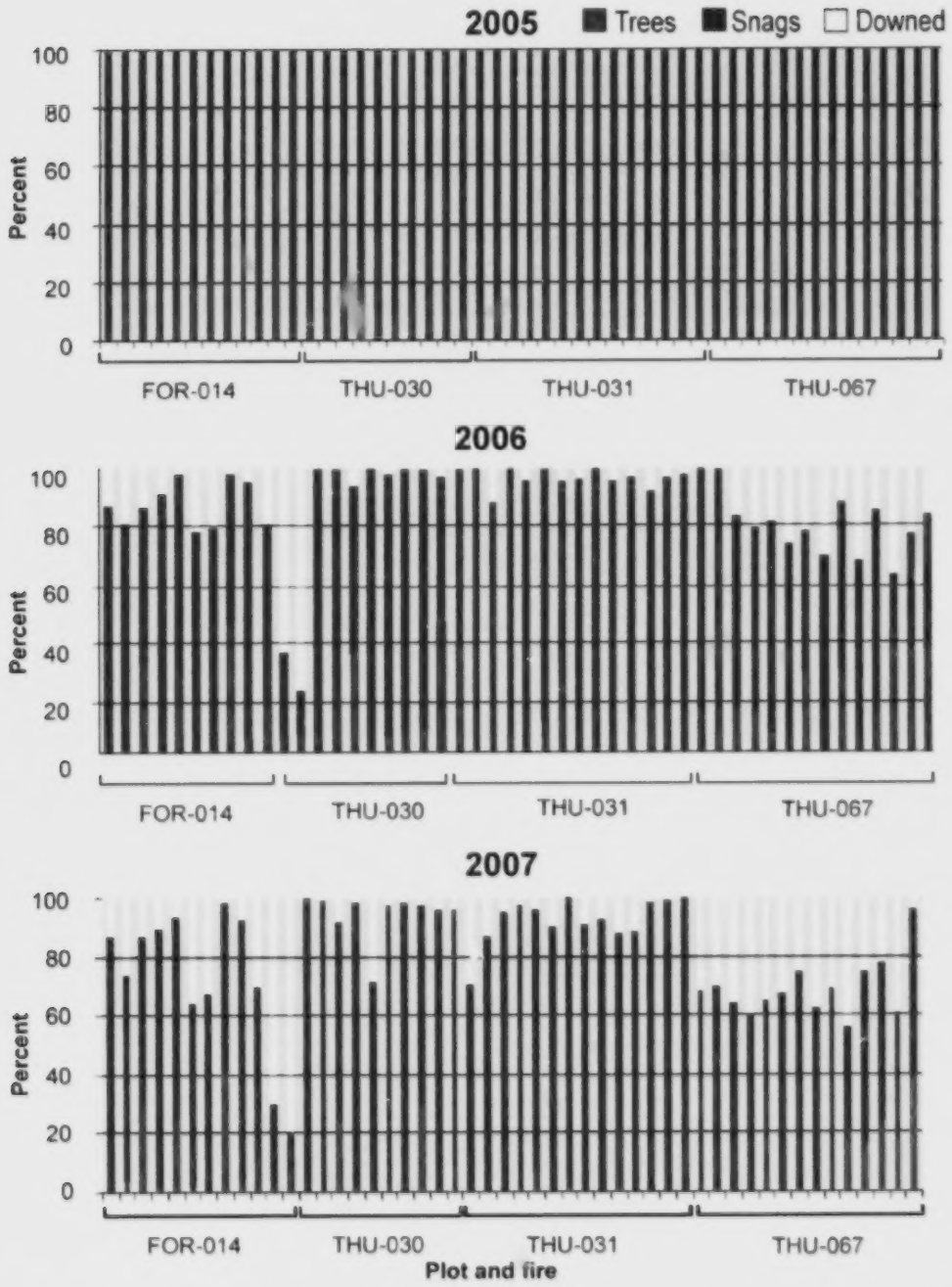


Figure 33. *Changes in the relative abundance of residual trees and snags in the field sample plots over three years post-fire. (a) Proportion of residual trees and snags immediately after fire (2005), (b) proportion of residual trees, snags, and downed stems – created by tree and snag fall after fire – in the second year (2006), and (c) proportion of residual trees, snags, and downed stems – created by tree and snag fall after fire – in the third year (2007).*

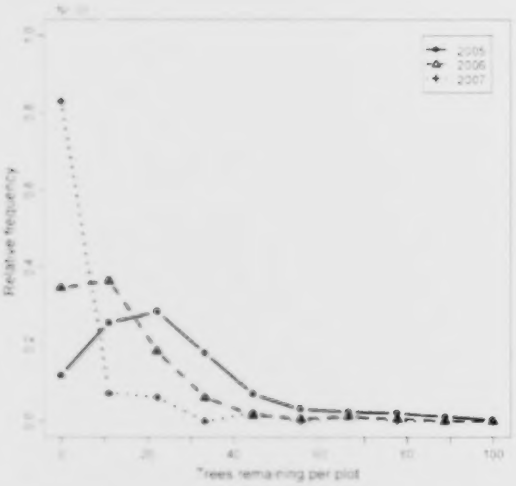


Figure 34. *Changes in the frequency of sample plots with a given number of residual trees during the first three years after fire across all 50 field sample plots in four fires. Probabilities should be read only at the points, not using the connecting lines, which are provided for illustrative purposes only.*

Another aspect of the changes in residuals is their relative frequency of occurrence across the monitored plots. As Figure 34 illustrates, the most striking change was observed in residual trees. The frequency of plots with residual trees became rarer with time; for example, the frequency of plots with no residual trees increased to 0.83 in the third year (from 0.12 in 2005).

The above discussion illustrates overall changes in abundance of the individual residual categories measured across 50 field plots in four boreal fires. Figure 35 quantifies the changes in abundance for each category of residual and illustrates the increase in the number of snags and in the amount of downed wood as residual trees die and fall and as snags fall. The reader is reminded that the field sample plots were biased to include plots that contained residual trees immediately after the fire, so the amount of residual structure reported here may not reflect that given for all sample plots (many of which had no residual trees to begin with, as discussed in the section on residual abundance).

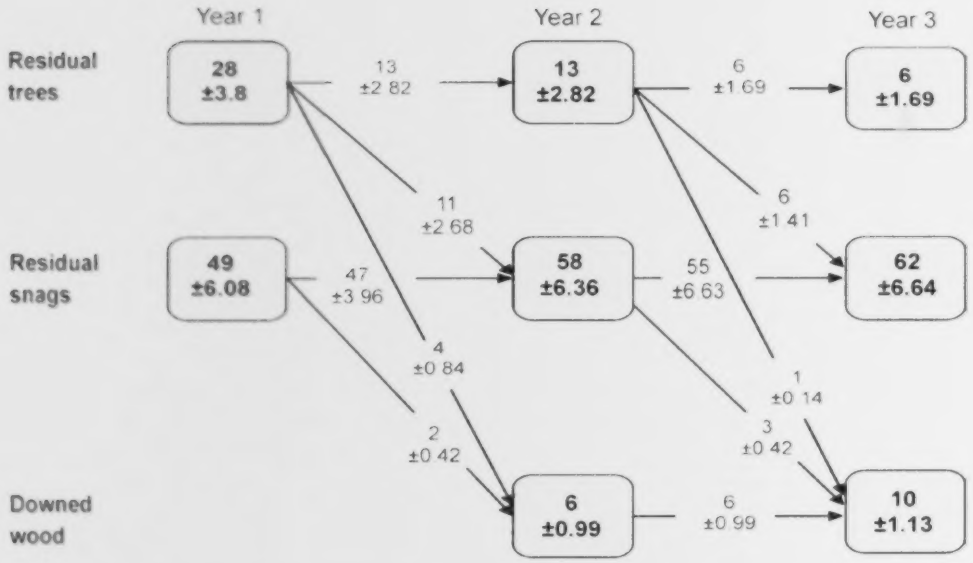


Figure 35. *Changes in the per-plot abundance (mean ± std. error) of residual trees and snags across 50 field plots in four study fires over the first three years after fire.*

Residual trees

Figure 36 illustrates the fate of the residual trees that remained immediately after fire (in 2005) over time. Of the residual trees present immediately after fire, 56% died and/or fell in the first year following the fire, and an additional 31% died and/or fell in the following year, leaving only 13% of those alive in 2005 still alive and standing in the third year. This figure also shows that most of the reduction in residual trees is due to mortality (67%), rather than tree fall (20%). As a result, more snags than downed wood are created by residual trees within the first three years after fire. Some examples of residual tree fall in the field plots are shown in Figure 37. Residual trees fell both in groups and individually.

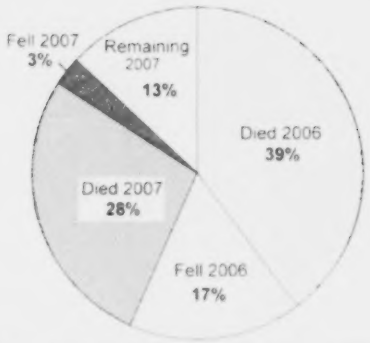


Figure 36. *Proportion of the residual trees that were present in 2005 that died or fell during the second and third year after fire in 50 field sample plots across four fires in boreal Ontario.*

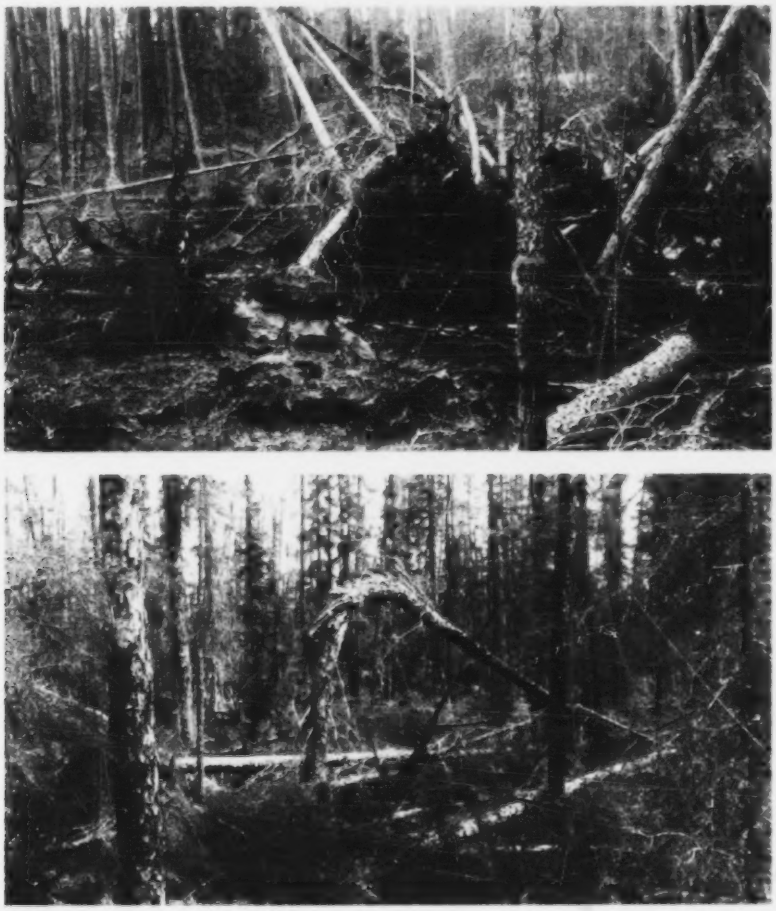


Figure 37. *Examples of residual tree fall in field plots.*

The proportion of residual trees that remained in a given year differed among the fires, but all four study fires showed the same dramatic drop in the proportion of residual trees remaining by year three (Figure 38).

Our results are similar to previous reports that expected survival of residual trees after fire is less than five years (Ohmann and Grigal 1979, Hely et al. 2003, Stambaugh 2003, Lavoie 2004) and three years (Viereck and Dyrness 1979, Haeussler and Bergeron 2004). When we scaled our observations of third-year residual tree results to a per-hectare level (by multiplying the sample plot totals by 20), the resulting values were comparable to the per-hectare abundance estimates presented in the literature for other boreal fires (Figure 39).

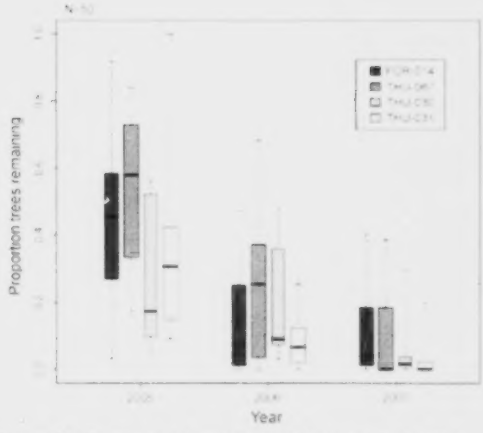


Figure 38. *Proportion of residual trees (relative to the total number of stems, i.e., trees+snags) remaining in each of four study fires in boreal Ontario for the first three years after fire.*

Figure 39. *Mean numbers of residual trees per hectare in the third year after fire for the four study fires, compared with values mentioned in the literature. (Colours indicate years after fire that data were collected: blue=1-3, grey=3-5.)*

Estimated distribution of residual tree abundance in the third year

Similar to the approach adopted to describe the per-hectare abundance of residuals immediately after fire, we bootstrapped (Davison and Hinkley 2006) the per-plot residual tree abundance in the third year to estimate the probability of per-hectare abundance. This was done by randomly selecting 20 plots from the overall set of 660 sample points, and an additional set of 20 randomly selected plots from the 50 field plots. From the first set (selected from the 660), we recorded the abundance of trees, snags, and downed wood in each of the 20 plots. From the second set of 20 (selected from the field plots), we calculated the proportion of trees that remained after three years, and the proportional increase or decrease in the number of snags and in the volume of downed wood. This yielded 20 sets of rates (one set per plot), which were then multiplied by the tree, snag, and downed wood abundances in the first set of 20 random samples from the overall HRP data. Thus, one bootstrapping replicate contained one estimate of the per-hectare abundance of residuals three years after fire. The hectare-level variability in residuals was calculated at the end of the simulation, using the entire set of 10,000 such estimates. The distribution that emerges represents a robust estimate of hectare-level variability and abundance of residuals, assuming no spatial dependence between plots.

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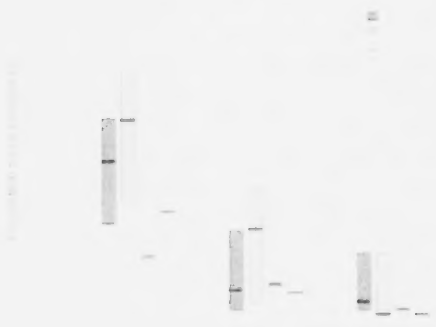

Figure 38. *Proportion of residual trees (relative to the total number of stems, i.e., trees+snags) remaining in each of four study fires in boreal Ontario for the first three years after fire.*

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A central assumption of this approach is that the sample points are independent, both of each other and with respect to their initial and temporal rates of change. We know *a priori* that these assumptions are at least partially false. Large-scale autocorrelation in many landscape patterns virtually guarantees that neighbouring plots are subjected to some similar dynamics, although the strength of the influence of these processes on residuals is not known. Furthermore, since fire intensity controls both initial and long-term transition rates in residuals, these patterns are also likely to be correlated. However, without knowledge of how these correlations influence residuals, bootstrapping provides a neutral approach to exploring spatio-temporal dynamics and gaining tentative answers to questions that would be impossible to answer otherwise.

Figure 40 shows the bootstrapped abundance of residual trees, expressed as the estimated probability of abundance values per hectare. It illustrates the probability distribution of expected abundance of residual trees per hectare, both as a histogram (for which the bar width shows the range of abundance and bar height shows the relative frequency of that range in the 10,000 bootstrap replicates) and as a cumulative probability curve (showing the relative frequency of occurrence of all values less than a specified point on the horizontal axis). The highest probability for any range was for 10 to 20 residual trees to occur per hectare three years after fire (relative frequency=0.3). In the third year after fire, more than 30 trees per hectare would occur in fewer than 20% of cases.

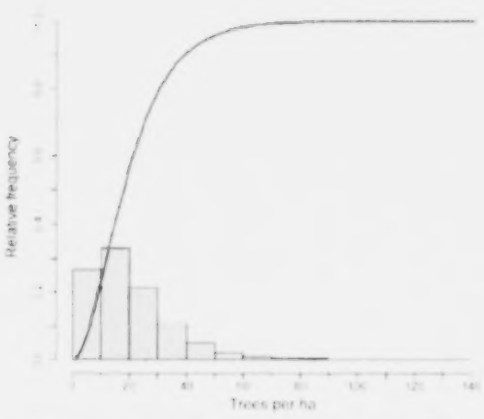

Figure 40. *Probability of estimated distribution of residual tree abundance per hectare three years after fire, shown as a point probability for ranges (histograms) and as a cumulative probability distribution (line).*

We further separated the estimated probability of the abundance of residual trees by diameter class to determine whether abundance varied with tree size. The estimated abundance of residual trees for the smaller diameter class (10.0-24.5 cm) was similar to the overall expected abundance (Figure 41a). However, the estimated abundance of residual trees for the larger diameter class (>24.5 cm) was much lower. The probability of more than one large-diameter residual tree occurring per hectare three years after fire is less than 0.3 (Figure 41b). This probability decreases to less than 0.1 for more than three large-diameter residual trees.

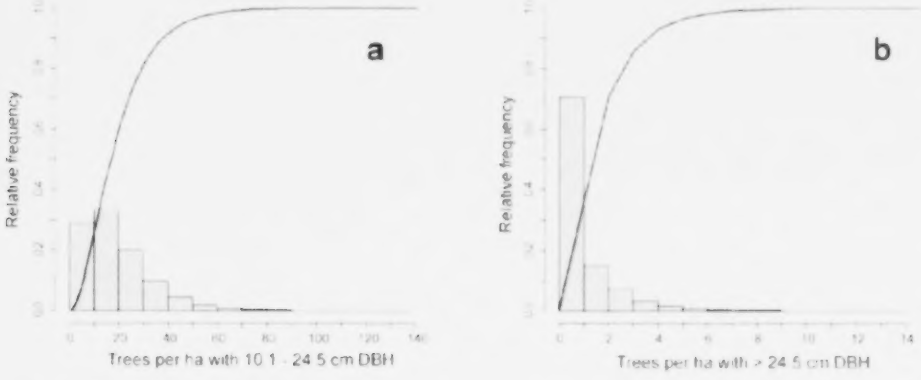

Figure 41. *Probability of estimated distribution of residual tree abundance per hectare three years after fire by diameter class (a=10-24.5 cm, b=>24.5 cm), shown as a point probability for ranges (histograms) and as a cumulative probability distribution line).*

Residual snags

Figure 42 illustrates the fate of the residual snags that were present immediately after fire (in 2005) over time. Of the residual snags monitored in the 50 field plots, only 11% fell in the three years after fire, indicating that most remained standing at least for the first few years after fire, adding little to the downed wood component.

The proportion of residual snags remaining in a given year differed among fires (Figure 43). Fall rates were gradual, with a greater proportion of snags remaining in fires THU-030 and THU-031, relative to THU-067 and FOR-014.

Similarly, reports in the literature indicated that few snags fall within three to five years after fire (Ohman and Grigal 1979, Viereck and Dyrness 1979, Apfelbaum and Haney 1981, Hansen 1983, Lee and Crites 1999, Nalder and Wein 1999) and most fall between five and 15 years after fire (Hobson and Schieck 1999, Schaeffer and Pruitt 1991, Sander 2003, Awada et al. 2004, Capar 2004, LeGoff and Sirois 2004).

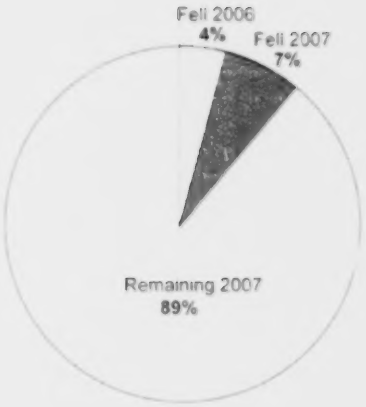

Figure 42. *Proportion of residual snags that were present in 2005 that fell in the second and third years post-fire in 50 field sample plots across four fires in boreal Ontario.*

Estimated distribution of residual snag abundance in the third year

Similar to the approach described for the residual trees, we bootstrapped the per-plot abundance of residual snags in the third year to estimate their per-hectare abundance. Figure 44 shows the bootstrapped per-hectare abundance of residual snags, expressed as probability distribution, both as a histogram and as a cumulative probability curve. The highest probability for a given range of abundances was for 900 to 1,000 residual snags to occur per hectare three years after fire (relative frequency=0.27). There is an extremely low probability (<0.0001) that no snags will occur in a hectare three years after fire.

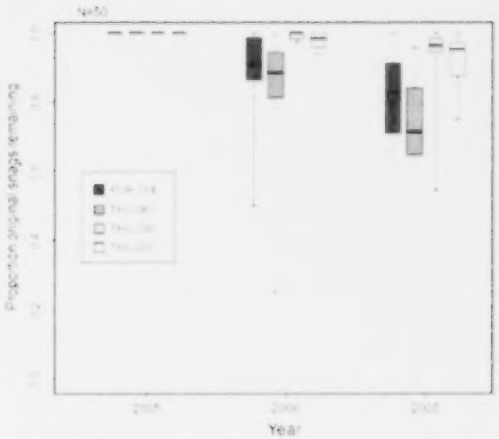

Figure 43. *Proportion of the residual snags (relative to the total number of stems, i.e., trees+snags) remaining in each of four study fires in boreal Ontario for the first three years after fire.*

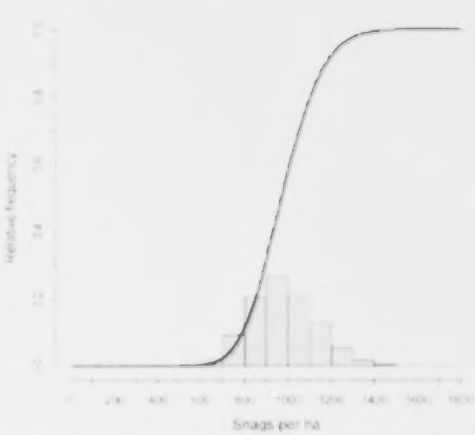

Figure 44. *Probability of estimated distribution of residual snag abundance per hectare three years after fire, shown as a point probability for ranges (histograms) and as a cumulative probability distribution (line).*

As with residual trees, we further separated the abundance of residual snags by diameter class to determine whether abundance varied with stem size. The estimated probability of residual snag abundance for the smaller diameter class (10.0-24.5 cm) (Figure 45a) was similar to but somewhat lower than the overall expected abundance. However, the estimated probability of residual abundance for the larger diameter class (>24.5 cm) was much lower. The probability of 125-150 large-diameter residual snags per hectare occurring three years after fire is 0.5 (Figure 45b).

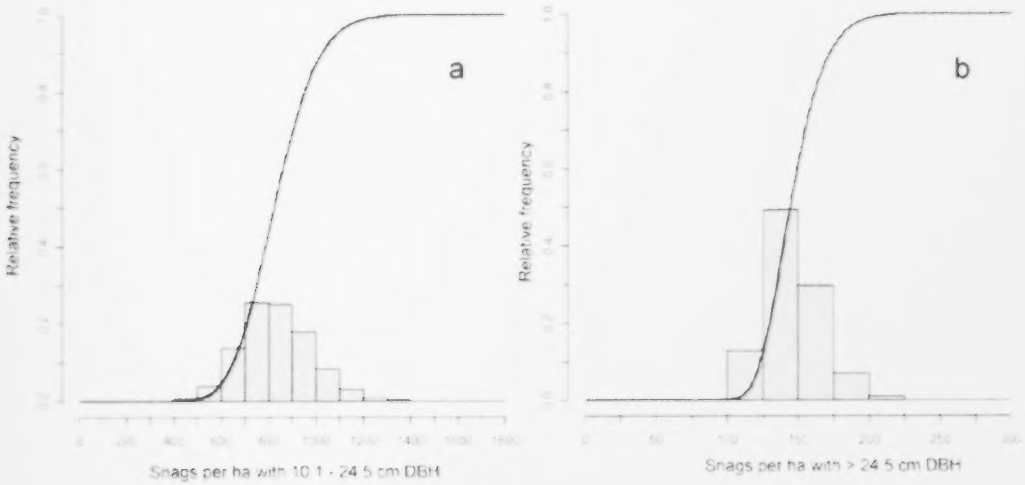

Figure 45. *Probability of estimated distribution of residual snag abundance per hectare three years after fire by diameter class (a=10-24.5 cm, b=>24.5 cm), shown as a point probability for ranges (histograms) and as a cumulative probability distribution (line).*

Conclusions

Our goals in this section are three-fold. First we summarize our estimates of residual abundance and their variability in comparison with estimates from published literature. Second, we discuss the results of our hypothesis testing for the sources of variability in residual abundance in comparison with results in the literature. In both instances, we used a literature review (Perera et al. 2007) that examined many reports of the abundance, occurrence, and spatial associations of stand-scale residuals from boreal North America. Finally, we compare our findings with the policy directions provided in OMNR's *Forest Management Guide for Natural Disturbance Pattern Emulation* (NDPE guide, OMNR 2001). We do so in the context of our definitions of components of stand-scale residuals and the scope of our study. One important distinction is that we focused on residuals that occur outside unburned patches. Our study results do not address contiguous areas within fires that remain unburned. Therefore, our results are relevant to findings and observations in literature only in that context and to the NDPE guide directions on residual tree retention specifically.

Residual abundance and variability

Residual tree occurrence was not common in the study fires immediately after fire. Overall, more than half of the plots had no residual trees, and in most of the study fires at least a quarter of the plots did not contain residual trees. Only rarely did plots have large numbers of residual trees, and as a result, the average abundance of residual trees was low, with only 8 per plot. However, the abundance of residual trees was not consistent across our samples, and varied widely both within and among study fires. We estimated that the highest probability (0.6) was to encounter 100 to 200 residual trees per hectare, immediately after a boreal forest fire, and there was a low probability (<0.2) of finding more than 200 residual trees per hectare. However, these probabilities changed dramatically soon after fire. By the third year, we found that the highest probability (0.32) was to find only 10 to 20 residual trees per hectare, and there was a low probability (<0.2) of encountering more than 30 trees per hectare. Most of the residual trees one would encounter in the third year post-fire were relatively small (<25 cm in diameter), and the probability of encountering larger residual trees (>25 cm in diameter) was very low: less than 0.3 for more than 1 residual tree, and less than 0.01 for more than 6 trees per hectare.

Residual snags were omnipresent in our study fires, with an average count of 33 snags per plot. Not only were there more snags than residual trees in almost all plots, their variability both within and among fires was also lower. We estimated that the highest probability (0.4) was to encounter 600 to 700 snags per hectare immediately after a boreal forest fire, and that it was less probable (<0.1) to find fewer than 500 snags. Unlike residual trees, the number of snags increases with increasing time after fire due to delayed mortality of residual trees and low fall rates for snags. Consequently, there was a higher probability (0.67) of encountering more snags (900-1,100 per hectare) in the third year after fire; it was less probable (<0.1) to find fewer than 600 snags in a hectare. There was also a higher probability (>0.75) of finding more, larger snags (150-175 per hectare; >25 cm in diameter) than residual trees.

As mentioned above, we also found that the abundance of residuals is time-dependent, that is, the abundance of residual trees and snags observed in a fire depends on the time since the fire event that created those residuals. Overall, the abundance of residual trees decreases sharply during the first three years after fire, and the number of snags increases as a result. As well, both trees and snags contribute to the residual downed wood during this period. Most significant are the changes in residual trees: more than 65% of the residual trees observed in the first year die to become snags, and a further 20% fall to become downed wood. In comparison, only 11% of snags fell during the first three years after fire. Over three years, residual tree mortality adds a further 35% to the number of snags that occurred immediately post-fire. Consequently, the contribution of residual trees to total residual abundance is only 8% of the total residual pool by the third year (in comparison to 38% in the first year). Therefore, estimates of residual abundance must be reported and considered in the context of the elapsed time since fire.

We are not confident in drawing conclusions about residual downed wood in our study due to the high error rates associated with its detection using airborne high-resolution photography. However, we can qualitatively note that all the fires we studied contained abundant downed wood, and that the amount varied both within and among fires. By monitoring the residual dynamics during the three years after fire, we estimated that an average of 10 stems per 0.05-hectare plot became downed wood as a result of the fall of residual trees and snags.

Comparisons of our estimates of residual abundance with those from other reports on post-fire residuals from boreal forest fires were difficult for several reasons. Some reports were one-time surveys conducted many years after fire events, so comparing their results to ours would be meaningless due to the differences in elapsed time. As well, the literature is replete with *post-hoc* and anecdotal observations that are not always distinguished from the results of rigorous analyses. Therefore, we compared our results with those available in the literature (see Perera et al. 2007 for details) by (a) selecting studies conducted within five years after fire and (b) separating study results from anecdotal observations. Even then, when trying to compare our study results with literature, it must be remembered that even in the best case in the literature (a) definitions of residuals were not consistent among studies, and (b) few researchers studied more than one fire, therefore did not report variability, thereby limiting the inference space for their results. We know from our study, with 11 fires, that fire-to-fire and within-fire variability are considerable. Within the context of these limitations, we found that our estimates of residual tree and snag abundance fell within the range of results reported by others, and the fire-to-fire variability observed in our results was also within the ranges reported by others.

Sources of variability in residual abundance

In addition to estimating the abundance of post-fire residuals, we examined their variability and the sources of that variability. At the outset of this study, we hypothesized that three major factors would affect the abundance and variability of residuals: the pre-burn forest cover, site conditions, and fire intensity. As in the simplified Figure 46, we acknowledged that these factors interact in their effect on residuals (for instance, the effect of fire intensity on some species may depend on the site condition) and/or may be spatially correlated (for instance, certain species may occur only on specific site types).

However, we could not examine all interactions and confounding factors among the different sources of variability for several reasons. First, the factorial combination of sources and the levels within each source would require more samples than we had available (even with 660 plots). Second, some sources are hierarchically nested (for example, fires nest cover types, which in turn nest site factors), and this required a balanced placement of samples across a range of those factors to meet statistical requirements, which was not possible in our sample set. Third, effects of many of these sources can be confounded (for example, cover types will affect fuel flammability, thus fire intensity, and have intrinsic tolerance levels to fire damage) and/or correlated (for example, certain cover types will occur only under specific site conditions), which further complicates comparisons of the interactions. As well, we could not use common statistical tests such as general linear models without violating their assumptions. The statistical tests we used (non-linear mixed-effects models) required more than the available number of samples (660) to examine interactions with an acceptable level of statistical power (for details, see Perera et al. in review-a). For these reasons, we could examine only their gross effect as sources of variability for residual abundance, one factor at a

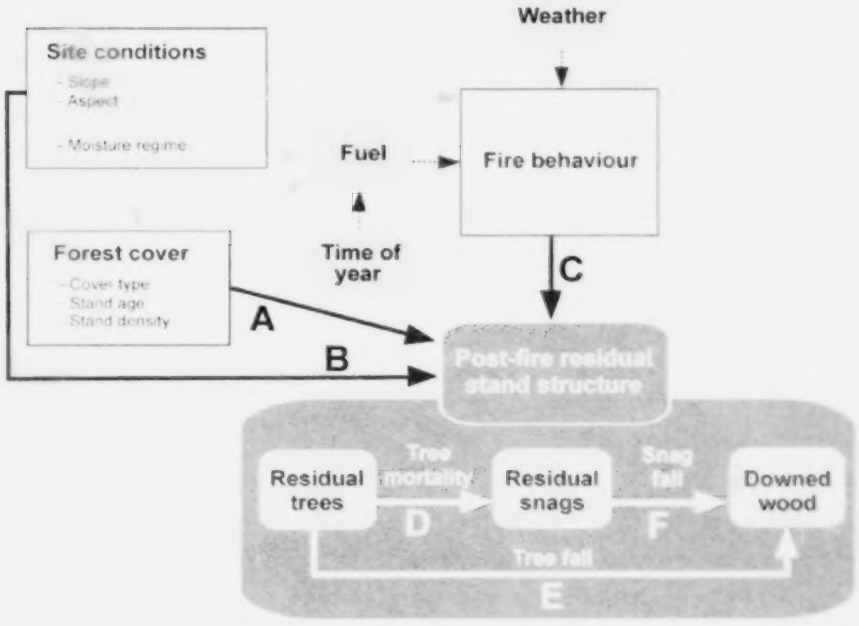

Figure 46. *A schematic overview of the major factors that influence post-fire abundance and variability of stand-scale residuals. Bold lines illustrate the gross effect of each factor, and dotted lines indicate interactions, confounding factors, and effects that we did not directly examine in this study.*

time. We recognize this results in an oversimplified portrayal, and may hide true patterns and/or reveal false patterns. As well, we acknowledge that the available data sources for forest cover and age (i.e., forest resource inventory) and soil moisture conditions (i.e., Ontario Land Inventory) may be too coarse and even inaccurate at the study plot level.

Variability in post-fire residual trees is often related to characteristics of the dominant forest cover type, average tree diameter, and stand age (see Perera et al. 2007 for details). In this study, we could not support the hypothesis that pre-burn forest cover characteristics have any direct effect (path A in Figure 46) on post-fire variability in residuals. As well, the hypothesis that moisture regime and terrain directly influence the occurrence and variability of residuals (path B in Figure 46) was not supported by our results. Although specific study goals, methods, number of fires sampled, and sampling resolutions may differ among the reports in the literature, we believe that the main reason for our inability to support these hypotheses resulted from our large sample space. Our number of sample fires (11) and sample points (660) far exceeded the highest number of study fires (4) and sample points (91) in the literature on post-fire residuals, and therefore would only detect influences if they were global, and outweighed the high variability both within and among fires in our study.

Variability in post-fire residuals was also attributed to differences in fire intensity in the literature, either by a proxy (e.g., distance to fire edge) or directly. Our results supported the hypothesis that high fire intensities significantly influence the variability in residual trees, regardless of the forest cover type, site conditions, and other variables (path C in Figure 46). This was evident in the effects of both distance-to-fire edge and in the local fire intensity. Therefore, we believe that fire intensity is the major driver of the occurrence of post-fire residual trees in fires, outside patches, and proposed a generalized hypothesis to explain the occurrence of residual trees immediately after fire (see Perera et al. in review-a): above a certain threshold of fire intensity (2,000 $kW \cdot m^{-1}$), all trees die, regardless of other associated factors; whereas below that threshold, post-fire residual trees are likely to occur, but their specific abundance and variability can be explained only locally, based on many interactions among (and confounded effects of) pre-burn forest cover, site conditions, and other factors.

No reports of specific studies on post-fire dynamics of residuals in boreal forests were found in the literature. However, the hypothesis that the post-fire abundance of residuals would vary with time, with residual trees dying and falling (paths D and E in Figure 46), and snags falling (path F in Figure 46), were supported by our observations during the first three years after fire. Temporal changes in the abundance of stand-scale residuals are therefore a key source of variability (Perera et al. in review-b), which makes comparisons of the results from different residual studies difficult or even irrelevant.

Study results in relation to NDPE guide directions

Below we relate our study results to the specific directions for residuals provided in the NDPE guide (OMNR 2001). We remind the reader that our study focused on stand-scale residuals that occur in burned areas within fires, and these results should not be confused with those for residual trees in unburned residual patches.

Guidelines for residual live tree and snag retention

Individual residual live trees and snags

• retain species based on fire tolerance, silvicultural requirements and wildlife habitat value (***guideline***)

Our results did not show any significant differences among forest cover types in the residual trees either immediately after the fire or three years later. We suggest that when fire intensities are low, many other factors interactively determine the abundance of residual trees, thereby masking the effects of individual factors such the relative fire tolerances of different species. When the fire intensities are high, all residual trees appeared to die, regardless of species differences. Therefore, our findings do not support the directions provided for fire tolerances of species in the NDPE guide as a global conclusion for boreal forest fires.

• *retain 25 well spaced trees/ha (minimum average) at least 6 large diameter, live, existing (or potential) high quality cavity trees (**standard**)*

The expected abundance of residual trees immediately after fire was much higher (approximately 150 per ha) than in the NDPE guide directions. However, within three years after fire, the abundance of residual trees decreased sharply, and fell within the guide directions. The probability of finding more than 6 large (>25 cm in diameter) residual trees, three years after fire, is extremely low (<0.01). Therefore, our findings suggest that the NDPE guide direction of leaving 6 large live trees is adequate, if they last up to three years. The post-fire abundance of snags far exceeded the NDPE guide directions; the estimated probability of finding more than 25 snags per ha approached 1.0 both immediately after fire and three years later, even for those with large diameters. We have no conclusive evidence about the spatial distribution of residual trees or snags within fires in relation to site conditions. In terms of their location and spacing, snags are omnipresent and thus can be left anywhere and everywhere; residual trees are more likely to occur where fire intensity is lower, such as near the edges of fires, and thus are best left where fires are not likely to be intense.

• range of tree species, diameters and condition (snags to healthy trees) to be left based on species (***guideline***)

We encountered considerable variation in the species, sizes, and conditions of residual trees and snags within and among fires. Therefore, our findings support the NDPE guide directions to leave a range of species, sizes, and conditions of residuals after disturbance.

Given the high variability within and among fires as well as over time, it is misleading to generalize post-fire residual abundance as per-hectare means. The probability of expected abundance is more informative and better captures the inherent variability in abundance. Though this variability has been attributed to pre-fire forest cover and site conditions in the literature, the sources of variability are difficult to isolate, even with the large sample size in our study. This is because the effects of these sources may be observed only locally, as may be seen with fewer samples. Beyond local scales, their effects may be confounded by interactions and cannot be generalized, especially with rigorous significance testing. The best generalized indicator of residual abundance appears to be fire intensity, which integrates the effects of forest cover and site conditions on post-fire residual formation. As well, the abundance and type of post-fire residuals change significantly over time; residual trees decreases, whereas those of snags and downed wood increase. Thus, any consideration of residual abundance must consider the elapsed time since the fire.

Guidelines for downed wood retention

Downed woody debris

- provide coarse downed woody debris through: using cut to length or tree length harvest systems; individual tree and snag retention; leaving unmerchantable logs on site; redistribution of roadside slash/chipping waste and avoid windrowing of coarse wood debris during mechanical site preparation (***guideline***)

- provide fine woody debris on shallow or very shallow or very coarse textured soils through: avoiding full-tree harvesting on these sites and redistribution of logging slash after roadside delimbing or chipping (***guideline***)

Relating our study results for the post-fire abundance of downed wood to the NDPE guide direction is difficult because the direction is very general. Although our abundance estimates for downed wood immediately after fire are not accurate, downed wood was ubiquitous after fire, and its abundance appeared to increase with increasing time since fire due to the fall of residual trees and snags. Therefore, our findings support the general direction in the NDPE guide to leave downed wood throughout disturbances.

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Appendix I. Details of the study fires

The following brief descriptions of the study fires have been summarized from MNR district fire reports. Specific fire parameters are provided in Table A-1.

NIP-020

The largest of the study fires, NIP-020 was ignited by lightning during warm, dry weather and a high to extreme fire index. Even though it was a summer fire and the trees had greened up, much of the area had recently been harvested, the mixedwood stands in the area contained up to 50% dead balsam fir, and part of the area had experienced blowdown during a windstorm in 2001, all adding to the available fuel. Most of the fire growth occurred over 2 days, with a head fire intensity ranked at 4 and classified as an intermittent or continuous crown fire during peak burn periods.

THU-030

This fire ignited under high to extreme fire weather indices, and was a summer fire in a boreal spruce fuel type. Most of its growth occurred within 3 days. Because it fell within Wabakimi Park, it was left to burn itself out.

THU-031

This fire ignited under high fire weather indices, and was a summer fire in a boreal spruce fuel type. Most of its growth occurred within 4 days.

THU-067

This fire ignited under high fire weather indices, and was a summer fire in a boreal spruce fuel type. Most of the fire growth occurred over 4 days, with a head fire intensity ranked at 4 to 5. It was classified as an intermittent or continuous crown fire during peak burn periods.

COC-010

This fire ignited under extreme fire weather indices, and was a spring fire in a boreal spruce fuel type. Most of its growth occurred within 2 days, with a head fire intensity ranked at 5. It was classified as a crown fire.

TIM-005

This fire ignited under high fire weather indices, and was a spring fire in a mature jack pine fuel type. Most of the fire growth occurred within 1 day, with a head fire intensity ranked at 4. It was classified as an intermittent or continuous crown fire during peak burn periods.

TIM-007

This fire ignited under high fire weather indices, and was a spring fire in a mature jack pine–deciduous mixedwood fuel type. Most of its growth occurred within 1 day, with a head fire intensity ranked at 5. It was classified as a crown fire.

TIM-019

This fire ignited under extreme fire weather indices in jack pine slash in a cutover, and was a summer fire that burned into an adjacent mature spruce forest. It was the second-largest of the study fires, and most of its growth

Table A-1. *Detailed fire information for the 11 study fires (based on data from MNR's Daily Fire Operations Support System, DFOSS).*

Eco-region	MNR fire code	Fine Fuel Moisture Code[1] (FFMC)	Duff Moisture Code[1] (DMC)	Drought Code[1] (DC)	Initial Spread Index[1] (ISI)	Build Up Index[1] (BUI)	Fire Weather Index[1] (FWI)	Fuel Type[1]	Rate of Spread[1] (m/min)	Fire Intensity Rating[1]
3W	NIP-020	91.2	17.8	81	12.6	52.8	26.8	M250 – Boreal mixedwood - green	20+	Extreme
	THU-030	91.5	48	168	4.1	51.5	11.4	C2 – Boreal spruce	6-10	High
	THU-031	91.5	48	168	4.1	51.5	11.4	C2 – Boreal spruce	6-10	High
	THU-067	84.6	23.5	228.8	4.3	54.9	12.4	C2 – Boreal spruce	11-20	High
	COC-010	95	42	101	19.7	42	33	C2 – Boreal spruce	11-20	Extreme
3E	TIM-005	91.5	35.7	120	7.4	41	16	C3 – Mature jack pine	6-10	High
	TIM-007	92	35.5	113.5	6.6	35.6	13.7	M175 – Boreal mixedwood - leafless	6-10	High
	TIM-019	90.1	72.6	394.1	7.8	99.5	26.4	S1 – Jack pine slash	6-10	Extreme
	CHA-018	91.9	85.2	359.9	10.7	107.1	34.1	C4 – Immature jack pine	6-10	Extreme
4W	FOR-014	89.5	39.4	257.3	5.1	52.9	13.8	M275– Boreal mixedwood - green	0-1	High
	FOR-024	89.8	24.8	317.7	3.8	35.9	8.8	M275– Boreal mixedwood - green	2-5	Moderate

[1] Based on the Canadian Forest Fire Behaviour Prediction System (Forestry Canada, Fire Danger Group 1992). FFMC: low, 0 to 80; moderate, 81 to 86; high, 87 to 90; extreme, >91. DMC: low, 0 to 15; moderate, 16 to 30; high, 31 to 50; extreme, >51. DC: low, 1 to 140; moderate, 141 to 240; high, 241 to 340; extreme, >341. ISI: low, 0 to 2.2; moderate, 2.3 to 5.0; high, 5.1 to 10.0; extreme, >10.1. BUI: low, 0 to 20; moderate, 21 to 36; high, 37 to 60; extreme, >61. FWI: low, 0 to 3; moderate, 4 to 10; high, 11 to 22; extreme, >22.

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